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Anticipation and Adaptation in Particulate Matter Policy:
The European Union, the Netherlands, and United States

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Anticipation and Adaptation in Particulate Matter Policy: The European Union, the Netherlands, and United States

1. Joint Introduction and Summary

The evolution of particulate matter (PM) air quality policy in the European Union and in the United States between 1970 and the present has been atypical. The US government and the European Commission have mandated scheduled reviews of PM policy over the past three decades and have updated that policy to new scientific information on multiple occasions. The use of planned adaptation over such a long period and in this manner, as a means to deal with uncertainty, has not often been reproduced in air quality policy.

Furthermore, particulate matter policy in the EU and US does not conform to the commonly held perception that the EU's environmental policies are, by and large, more precautionary than the respective policies in the United States. The US decisions to adopt air quality standards for PM_{10} and $PM_{2.5}$, in 1987 and 1997 respectively, led those in the EU by approximately nine years.[1] An analysis of the comparative stringency of the PM standards in the US and EU shows that the $PM_{2.5}$ standard the US implemented in 1997 is more stringent than the standards that have been proposed in the EU by the European Commission and the European Parliament. In September this year, the US repealed their annual standard for PM_{10}. Prior to that, however, the annual PM_{10} standard the EU implemented in 1999 was more stringent than the one the US adopted in 1987. The daily PM_{10} standards in the EU and US are of similar stringency. In the Appendix, these comparisons in stringency are discussed in more detail.

The differences between the EU and US policies are remarkable because they are based on the same science and therefore reflect dissimilar processes of interpreting that science and the uncertainties inherent in it. The two cases themselves focus on the science-policy interfaces for their respective governing bodies. The EU case also looks at the science-policy interface in the Netherlands. The US case also examines policies for sulfur dioxides that relate to the PM policies. The remainder of this summary discusses how characteristics of the science-policy interfaces may have led to the differences in outcomes.

[1] PM_{10} refers to particulate matter with mean aerodynamic diameter less than 10 microns and $PM_{2.5}$ refers to PM with mean aerodynamic diameter less than 2.5 microns. The EU decision to set standards for PM_{10} was made in 1996 when the EU air quality Framework Directive was adopted. The precise standards were set in 1999. The EU has not yet formally decided to regulate $PM_{2.5}$, but the European Commission proposed a standard in 2005 and the European Parliament proposed another standard in 2006. Whatever the precise outcome of the negotiation process between the Council of Ministers and the European Parliament, the decision has now effectively been taken to regulate $PM_{2.5}$. The US set PM_{10} standards in 1987 and set $PM_{2.5}$ standards in 1997. They recently made those $PM_{2.5}$ standards more stringent in September of 2006 and repealed one of the two PM_{10} standards in the US.

1.a. The difficulty of anticipation

The cases identify a number of factors that influence the success of anticipation and adaptation as means for dealing with uncertainty when developing policy. Similar factors influence both anticipation and adaptation. For example high-quality knowledge assessments increase the likelihood that both will succeed, as do incentives to surface information. The transparency of processes and models may also influence both.

Table I shows examples in which the costs, benefits, or effectiveness of policies were anticipated prior to the implementation the policy. As one would expect, attempts at anticipation of outcomes were not perfect – although the range of success varied considerably.

In the US Acid Rain Program, a lack of saliency in knowledge assessment contributed to the severe misanticipation of the program's benefits. In the US air quality standards cases, the benefits and costs are not estimated prior to implementation because the Clean Air Act does not allow their consideration. However one can say, for example, that the health benefits or effectiveness of limiting PM_{10} with an annual standard were less than expected because the Environmental Protection Agency (EPA) recently repealed that standard. The misanticipation in the case of PM_{10} was caused by the uncertainties in the science about PM's health impacts. Those uncertainties still exist, as scientists are not sure that particles between 2.5 and 10 μm in diameter have no effect on health. The EPA did not revoke its daily standard for PM_{10}, which is more stringent than was its annual PM_{10} standard. For similar reasons, both the US and EU stopped regulating total suspended particulates (TSP) when scientific studies showed that smaller particles were responsible for the majority of the PM's health impacts.

The case of building projects in the Netherlands provides an example where misanticipation was even more extreme. The Netherlands chose a more strict interpretation of the EU air quality limit values than did many other EU countries.[2] The Netherlands also required the use of detailed modeling to help predict the impact of construction projects on air quality. Policymakers did not initially consider the high economic costs of this strict interpretation of the EU air quality limit values, and they did not foresee the strict way in which the judiciary would actually enforce the air quality order of 2001. They are now reconsidering their method of interpretation of the standards. The misanticipation in this case was the failure to anticipate the high economic and social costs of the policy.

1.b. Examples of adaptation

These cases are examples of why a strategy of adaptation is needed as a means of dealing with uncertainty: the anticipation of the effects, benefits, and costs of policies are often incorrect. If adaptation does not occur, policies remain designed around initial, unavoidably incorrect, anticipation of outcomes. This can carry high social costs, either in the form of

[2] Sweden being the only EU country with an interpretation that was even more strict than that of the Netherlands.

excessive cost burdens (as in the case of building projects in the Netherlands) or in the form of forgone health benefits (as in the case of the Acid Rain Program in the United States). Anticipation will always be needed because even when adapting a policy, uncertainties will persist. These reasons make an iterative process a desirable outcome.

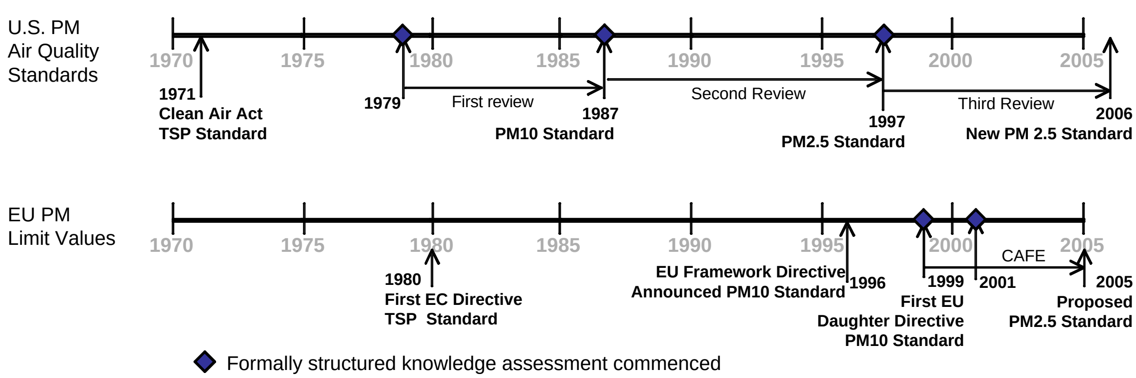
TABLE I: The location of the anticipated costs, benefits, or effectiveness of policies on a spectrum of expected to unexpected. If the effects where anticipated, they are also placed on a spectrum of overstated to correctly predicted (materialized) to understated.

	Expected			Unexpected
EXAMPLE CASE	**Overstated**	**Materialized**	**Understated**	
Acid Rain Program	Costs		Benefits	
U.S. TSP Air Qual. Stndrds*	Benefits			
EU TSP Air Qual. Stndrds*	Benefits			
U.S. PM10 Air Qual. Stndrds*	Benefits			
U.S. PM2.5 Air Qual. Stndrds*	Costs		Benefits	
Netherlands building prjtcs	Benefits			Costs

* Not explicitly calculated, but effectiveness of the policies were expected none-the-less

The comparison of the EU and US PM air quality cases reveals similarities and differences. Both have used adaptation to adjust policies as the science about the health impacts of particulate matter progressed. One difference between these cases is that the US led the EU in adopting new standards, as shown in the timeline in Figure 1. Another difference is that the US $PM_{2.5}$ standard is more stringent than the one the EU is currently considering. Another is that the EU annual standard for PM_{10} is more stringent than the annual standard for PM_{10} that was in place in the US until September 21, 2006.

Figure 1: Timelines of EU and US particulate matter decisions



1.c. Similarities between the EU and US air quality decisions for particulate matter

The US and EU rely on the same science to make decisions about air quality standards. This science includes both toxicological and epidemiological studies about the impact of PM on public health. The body of science has dramatically increased since the early 1990s. The epidemiological studies have provided the majority of evidence that there is a connection between the smaller particles and detrimental health impacts including things like asthma and cardiovascular disease. There still remains considerable uncertainty about the mechanisms of the connection between small PM and health effects that the epidemiological studies observe.

Beyond relying on the same body of science to support air quality decisions, the US and EU cases are similar in other important ways. For example, both use highly transparent, participatory processes to inform policy decisions about relevant science. In the EU case, formal procedures for reviewing the standards were put in place after the 1999 decision to implement the PM_{10} standard and the Clean Air for Europe (CAFE) assessment program started in 2001. The EU designed CAFE specifically around the goal of providing high quality science-policy advice through a transparent knowledge assessment process. The US process is also transparent, calls for public comment and input a number of times, and includes mandated and scheduled reviews of the standards and the science behind them. Interestingly, the EU implemented standards for total suspended particles, PM_{10}, and $PM_{2.5}$ about nine years after the US in each case. This suggests that a nine-year period is roughly how long these transparent, participatory review processes need to complete a review of an air quality standard.

1.d. Differences between the EU and US air quality decisions for particulate matter

The question about why there has been a mismatch in timing between the US and EU adoption of PM standards can be posed two ways. First, why have there been lags in the adoption of PM standards on the part of the EU? Second, why did the US interpret uncertainty more aggressively and adopt new PM standards more quickly than the EU?

One reason that may have caused the EU decisions to lag those in the United States is that during the 1990s the majority of the influential epidemiological studies were performed using cohort data collected in the United States. Decision makers in the EU could have felt that some aspects of these studies, like differences in the population, composition of the particulate matter, or other aspects of air quality were not representative of the EU. However, in 1987 the US decision to adopt PM_{10} standards relied in part on reanalyses of the London fog episodes in the 1950s. These reanalyses were more applicable in the EU than in the United States. The US Environmental Protection Agency used other US studies to "translate" the London fog studies to US conditions but they did rely on the London fog studies to help them set the upper bound for the standards considered in the 1987 review.[3] Other reasons for the relative slowness in the EU could have been politics that were external to decisions about air quality or the opposition of industries in some countries.

There are a number of reasons that could help explain why the US has been quicker to adopt new PM standards, given the uncertainties, than the EU. First, the Clean Air Act does not allow the EPA to consider the costs of adopting air quality standards that protect the public health. Second, as explained in the US case study, interest group lawsuits pushed the EPA to review the PM standard by a deadline each time they revised it. Although the EU did have a mandated, scheduled review procedure in effect since 2001 (the first review was completed by the European Commission in 2005), there was not a similar mechanism by which interest groups could have pushed the revision of standards, since the review was solely under the authority of the Commission.[4]

Incentives to surface information are another important influence on these processes. In the US, the desire of many different parties to see the US Acid Rain Program succeed created strong incentives for information about the health impacts of $PM_{2.5}$ and their connection to sulfur dioxide pollution to surface. The EPA, Congress, and others rely on the science backing the $PM_{2.5}$ air quality standards to maintain that the Acid Rain Program has been successful. The Acid Rain Program reduces some forms $PM_{2.5}$, like sulfates, through its reductions in precursor emissions (sulfur dioxide and nitrogen oxides). But, there is still some uncertainty about whether sulfates, for example, are the fraction of small PM that damage health the most. The US downplays this uncertainty because of incentives to claim that the Acid Rain Program has succeeded. The EU does not have a similar incentive, and does not downplay the uncertainty about which fractions of PM are the most detrimental to public health.

1.e. Conclusions

Setting aside the differences in timing between the US and EU policy decisions on PM air quality, the courses of PM policy on both continents are laudable simply because they use planned adaptation, and even unplanned or forced adaptation, to help deal with uncertainties. As is normal in environmental policy, uncertainties mean that we cannot be certain if the US is correct in focusing its attention on smaller and smaller PM particles or in believing in a connection between reductions in sulfur dioxides and harmful fractions of PM. We also cannot be sure whether the EU or the US has erred. Has the EU forgone health benefits by their comparative delays in the implementation of strict $PM_{2.5}$ standards? Or, has the US incurred unnecessary costs? We can only be sure that – because of uncertainties – both the US and the EU have made suboptimal decisions, to a small degree at least.

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[3] See the U.S. Environmental Protection Agency's "Criteria Document" cited in the U.S. Case Study as EPA CD 1982.

[4] The review was delayed twice. The deadline set in 1999 was 2003. When the review was subsequently taken up in the context of the CAFE program that started in 2001, the new deadline was 2004. The new proposal was only published by the Commission in 2005, however, again a year later.

The history and institutionalization of planned adaptation in these cases means that we can be fairly confident that these suboptimal decisions will be changed again as science and policy experimentation create new knowledge. The same cannot be said of other environmental policies that share a reliance on uncertain justifications but that have never, and likely will never or will rarely, be changed. Despite their imperfections, PM air quality policy on both continents provides an example worthy of emulation. The attached case studies address some of the challenges that the EU and US PM air quality science-policy interfaces confronted to achieve this success. We hope that further work on the case comparisons will also, eventually, provide more concrete lessons.

Adaptation and Anticipation in EU and Dutch Particulate Matter Policies

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2. Introduction

Ambient particulate matter is air pollution consisting of a complex mixture of particles of various diameters and various chemical compositions. Depending on the diameter of the particles, either the abbreviation PM_{10} is used (for particles with a diameter up to 10 micrometers) or the abbreviation $PM_{2.5}$ (for particles with a diameter up to 2.5 micrometers). Exposure to PM in ambient air has been linked to a number of different health outcomes, ranging from modest transient changes in the respiratory tract and impaired pulmonary function, through increased risk of symptoms requiring emergency room or hospital treatment, to increased risk of death from cardiovascular and respiratory diseases or lung cancer. This evidence stems from studies of both acute and chronic exposure, and from toxicological studies (see, e.g., WHO 2006). Because of these health risks, the regulation of ambient particulate matter has become part of air quality policies across the world. For instance, the US and EU are regularly updating their air quality standards for PM on the basis of progressing scientific evidence. And cities in Asia are beginning to implement command-and-control policies to reduce PM emissions from traffic and industry.

On the basis of scientific studies and models, governments prepare air quality policies to mitigate the potential health problems associated with particulate matter. The policies can be said to *anticipate* the actual presence of these problems. Anticipation is difficult, since the evidence is inconclusive and challenged in the societal debate. For instance, in the integrated assessment that precedes European environmental policy making, a diverse array of models is used from different disciplines, each having uncertainties attached to them. Also at the (sub)national level, the impact of concrete policy proposals on air quality is often assessed using models. In the Netherlands, air quality models are used, for instance, to assess the impacts of spatial planning projects. Since scientific and policy realities may turn out to be different from what was anticipated in the initial formulation of policies, a situation may arise in which it becomes desirable to *adapt* the policies to meet the progressing insights. Such adaptation of policies involves new anticipations, and the new policies may again turn out to be subject to subsequent revisions, etc. This marks the iterative character of science-based policy making.

In this paper, first an overview is given of recent developments of particulate matter policies in the EU in general and in the Netherlands, one of the EU countries, in particular (section 3). These developments, which are currently culminating in negotations of a $PM_{2.5}$ standard, are put in a larger historical context (section 3.a). Also the way the science–policy interface is structured in EU policy making on particulate matter is described (section 3.b).

The questions that prompted this case study – and its companion US case study – relate to the notions of anticipation and adaptation. This paper examines instances of different types of anticipation and adaptation in EU and Dutch decision making on PM (section 4). The following questions on *adaptation* are dealt with: In what ways can recent the policy change with respect to the introduction of a $PM_{2.5}$ standard be said to be a 'planned adaptation', that is, to what extent did previous EU policy decisions explicitly keep open the possibility that new scientific information available at a scheduled review date could lead to changes in PM standards, including the introduction of a $PM_{2.5}$ standard? (section 4.a.i.) What processes other than planned adaptation through scheduled reviews triggered the 2005 softening of the Dutch air quality order that implements the current EU Directive on PM_{10}? (section 4.a.ii) What causes the approximately 9-year delay in EU decision making on PM as compared with the US while the science base is the same? (section 4.a.iii).

Section 4.b focuses on *anticipation*. First, issues such as the transparency of the modeling process and creating credibility for modeling results are touched upon (sections 4.b.i). Subsequently, the assessment of knowledge and the treatment of uncertainties in policy making are dealt with in more detail, both at the EU level for the setting of standards (section 4.b.ii) and at the Dutch level for the implementation of the EU Directive regulating PM_{10} (section 4.b.iii).

This paper concludes with a brief discussion (section 5).

3. Background on EU and Dutch Particulate Matter Policies

3.a. Timeline of Key Decisions

EU policies on emissions and on air quality standards date from the 1980s, but the EU only became really active in the field of air pollution policy making in the 1990s. Before the 1990s, individual European countries regulated PM in different ways.[5] In this paper, we will only focus on one country in particular, the Netherlands.

[5] Still, EU countries are allowed to implement more stringent norms (see, e.g., Sweden). And, perhaps even more important for this study, different countries implement the EU Directives in different ways.

Policies concerning emissions of PM and PM precursors

1970 — In the Netherlands, the Wet inzake de Luchtverontreiniging (Dutch Clean Air Act, WLV) was introduced, a framework law that enables the regulation of polluting installations, fuels or activities. Examples of regulatory decisions under the WLV: categories of industries to which the WLV applies (1972); sulphur content of fuels (1974); standards for primary PM emissions from stoves and multiburners (1996).

1976 — Dutch national emission ceiling of 500 kton SO_2 (IMP Lucht 1976-1980).

1979 — UNECE Convention on Long-Range Transboundary Air Pollution.

1980 — The Wet milieubeheer (Dutch Environmental Protection Act, WM), which is currently the most important Dutch environmental law (also a framework law), was introduced. Examples of regulatory decisions under the WM: PM emissions from small industrial burners (1990); PM emissions from waste-burning facilities (1993).

1985 — UNECE Sulphur Protocol, Helsinki: all countries agreed to a 30% reduction of national SO_2 emissions.

1987 — EU directive regulating the emissions of pollutants, including PM, from diesel engines, laying the foundation for later directives (1999, 2001, 2005). Subsequent standards (increasingly strict) are introduced by this series of directives: EURO I in 1992, EURO II in 1996, EURO III in 2000, EURO IV in 2005 and EURO V in 2008.

1988 — EU Large Combustion Plant Directive, setting emission limits for SO_2, NO_x and dust for new and existing plants (revised in 2001, together with appearance of the EU National Emission Ceilings Directive, see below).

1994 — UNECE Second Sulphur Protocol, Oslo: application of effects-based approach; best available technology; application of economic instruments; and critical load concept.

1999 — UNECE Gothenburg Protocol (a multi-polutant, multi-effect protocol in which the mitigating effects on secondary PM are considered as 'additional benefits'), signed by the EU member countries but not by the EU itself.

2001 — EU National Emission Ceilings Directive, which regulates the emissions of several pollutants (including major PM precursors such as SO_2 and NH_3) and which is for some species more stringent than the 1999 UNECE Gothenburg Protocol.

2004 — Change in Dutch tax law to stimulate soot filters in diesel cars.

2006 — Example of another type of policy instrument than regulation or tax measures: Dutch covenant between national government, city governments and organizations from the transport sector on the introduction of 'environmental zones' within cities, which only 'clean' trucks are allowed to enter.

Policies concerning PM air quality standards

1980 — The first EC Directive on ambient air quality was adopted by the Council (80/779/EEC). Two norms for 'suspended particles' (all size fractions included) were introduced: at the end of 1983, the 98 percentile of all daily mean values taken throughout the year should not exceed 250 μg/m^3 and the median of daily mean values throughout the year should not exceed 80 μg/m^3. Member States must measure concentrations, and report any breaches of the limit values to the European Commission.

1996 — In 1996, a harmonisation process between EU countries began with the entry into force of the EU Air Quality Framework Directive (1996/62/EC), replacing directive 80/779/EEC, among others. This Framework Directive provides a new and coherent general European framework for 'evaluating and managing air quality'. The Framework Directive uses a number of important concepts: daughter directives, preliminary assessments, assessment thresholds and zones and agglomerations. The daughter directives are specifications of air quality requirements for certain substances. In the meantime, four daughter directives have appeared.

1999 — The first EU daughter directive regulates PM_{10} (air quality standard), among several other air pollution species (1999/30/EC), setting standards that should be met by 2005 and 2010. In 2005, the annual average EU norm is 40 μg/m^3 and the daily average EU norm is 50 μg/m^3, which may be exceeded 35 times at maximum. The daughter directive schedules a review for 2003, keeping open the possibility of adding a $PM_{2.5}$ norm. $PM_{2.5}$ should be measured and reported.

2001 — The first daughter directive is implemented in the Netherlands as an Air Quality Order (Staatsblad 2001) under the Dutch Clean Air Act in 2001.

2005 — The Dutch goverment issues a new Air Quality Order, replacing the one of 2001. This order contains a more flexible intepretation of the air quality directives: subtraction of the natural PM fraction; averaging of air quality over larger areas; no stand-still principle (allowing for increasing PM concentrations where they stay under the norm).

2005 — Communication from the European Commission "Thematic Strategy on Air Pollution", the result of the Clean Air for Europe (CAFE) programme and part of the Community's Sixth Environmental Action Programme. Aimed at streamlining existing provisions and merging five legal instruments into a single Directive. Proposal to introduce new air quality standards for fine PM ($PM_{2.5}$): $PM_{2.5}$ limit value of 25 μg/m^3 (annual average) to be attained in 2010. In the longer term, a 75% concentration reduction is proposed, which implies a reduction in primary $PM_{2.5}$ emissions by 59%.

2006 — On March 17, 2006, a legislative proposal was sent to Dutch parliament to integrate a large part, including the limit values, of the existing Dutch regulatory decisions implementing the EU Directives, such as the 2005 order, into the WM.

2006 — The proposed directive that includes a $PM_{2.5}$ norm is amended by the European Parliament. A more strict – but indicative instead of obligatory – target of 20 μg/m^3 (annual average) is proposed, with the explicit possibility to make it obligatory after review.

3.b. The EU Science–Policy Interface: CAFE program[6]

The question of how EU policies anticipate uncertain air pollution problems and adapt to progressing insights can only be answered by first taking a closer look at the science–policy interface. Here we will briefly discuss the EU science–policy interface with a focus on the role of models in the Clean Air for Europe (CAFE) program that led to the 2005 Thematic Strategy on Air Pollution (European Commission 2001a).

CAFE (see also box 1) was developed under the leadership of a permanent secretariat housed within the Directorate General Environment of the European Commission. A Steering Group composed of about 80 representatives of the Member States, the European Parliament, stakeholders and relevant international organisations meet two or three times a year. Its mandate was to advise the Commission on the strategic direction of the programme rather than on technical issues. It did not have any formal decision making power. Nor did any of the other groups in the CAFE program. The mandate of the CAFE program was the development of policy guidance, not of the policies itself. The CAFE program was situated in the first phase of the development of a European legislative policy proposal, the so called 'expert phase'. This is the phase during which the Commission collects information to develop a policy proposal.

Furthermore, according to the readers guide to the CAFE work plan (European Commission 2001b), 'the policy guidance to emerge from CAFE needs to be based on an integrated assessment of a wide range of policy alternatives, taking account of all relevant scientific, technical and political information'. The CAFE program organized its input and integrated assessment work through a Technical Analysis Group (TAG) and a Working Group on Target Setting and Policy (WG TSP). The Technical Analysis Group (TAG) consisted of members of the CAFE secretariat and consultants carrying out technical analysis under specific contracts. The group consists of about 10–20 people and met once a year. The technical analysis in CAFE was mainly carried out under these different contracts and therefore the role of the consultants was very important.

The TAG was mainly set up to enable the consultants to co-ordinate between themselves. Contracts included (1) Development of a Baseline Scenario and an Integrated Assessment Model (International Institute for Applied Systems Analysis, Austria; Meteorological Institute, Norway; National Technical University Athens, Greece) (2) Further Development and Application of the TREMOVE Transport Model (Catholic University, Leuven, Belgium) (3) Cost-Benefit Analysis of the CAFE Programme (AEA Technology) and (4) Review of Health Effects (World Health Organisation, Geneva). The consultants presented the progress of their work in the Steering Group meetings. All contractors had been involved in analysis for the European Commission already in earlier policy development processes. The RAINS model had already been used for the preparation of the

[6] This section is based on an article by Tuinstra, which is submitted to Environmental Science and Policy.

Box 1. Objectives of the CAFE Program

"Clear Air For Europe will have the general aim of developing a long-term, strategic and integrated policy to protect against the effects of air pollution on human health and the environment. As required by the treaty, the policy will aim at a high level of environmental protection based on the precautionary principle, taking account of the best available scientific and technical data and the costs of benefits of action or lack of action." (European Commission, 2001a)

The specific objectives of CAFE are (CAFE- website, 2005):

1. to develop, collect and validate scientific information relating to the effects of outdoor air pollution, emission inventories, air quality assessment, emission and air quality projections, cost-effectiveness studies and integrated assessment modeling, leading to the development and updating of air quality and deposition objectives and indicators and identification of the measures required to reduce emissions;
2. to support the implementation and review the effectiveness of existing legislation, in particular the air quality daughter directives, the decision on exchange of information, and national emission ceilings as set out in recent legislation, to contribute to the review of international protocols, and to develop new proposals as and when necessary;
3. to ensure that the sector measures that will be needed to achieve air quality and deposition objectives cost-effectively are taken at the relevant level through the development of effective structural links with sectoral policies;
4. to determine an overall, integrated strategy at regular intervals which defines appropriate air quality objectives for the future and cost-effective measures for meeting those objectives;

NEC directive, the Acidification Strategy and the Ozone Strategy by DG Environment, and the PRIMES energy model of the National Technical University Athens and the TREMOVE model had been used in earlier analyses for DG Transport and Energy.

The purpose of the Working Group on Target Setting and Policy (WG TSP) was to assist the Commission in the development of air quality related targets for the protection of human health and the environment. It also gave advice on issues related to policies and measures. It advised on the Integrated Assessment Modelling work and the choice of scenarios. The WG TSP met about four times a year and according to the membership list had 18 members: 13 country representatives of Environmental Ministries or Environmental Protection Agencies, a representative of the UN-ECE and four environmental and business Non Governmental Organisations.

Since the integrated assessment process described here took place in the 'expert phase' of EU legislation development, the Commission services, personified by the CAFE secretariat took the lead. The organisation of the process can be seen as being top-down. The civil servants of the Commission who developed the proposal played a very important role. The role of the scientists was one of being a consultant. The role of the countries (representatives in the steering group) was to give comments. In the steering group meetings there was no need to arrive at a consensus.

One of the key tasks in the CAFE programme was the development of a baseline scenario and an integrated assessment model. CAFE has compiled a set of baseline projections outlining the consequences of present legislation for the future development of emissions, of air quality and of health and environmental impacts up to the year 2020 (Amann et al. 2005). In further steps, the CAFE integrated assessment has explored the costs and environmental benefits associated with gradually tightened environmental quality objectives, starting from the baseline (current legislation) case up to the maximum that can be achieved through full application of all presently available technical emission control measures (the maximum technically feasible reduction case) (Amann et al., 2005).

The CAFE assessment is based on recent scientific knowledge, taking into account:

- Advice received from the World Health Organization on the health impacts of air pollution
- Information on vegetation impacts of air pollution compiled by the UN-ECE CLRTAP Working Group on Effects
- Syntheses of the understanding and modelling of the dispersion of air pollutants in the atmosphere at the regional scale developed by the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air pollutants in Europe (EMEP)
- Synthesis of the results of the so called City-Delta project, an open model inter-comparison exercise to explore the changes in urban air quality predicted by different atmospheric chemistry-transport dispersion models (CTMs) in response to changes in urban emissions. The range of response resulting from this model inter-comparison is to be used in the cost-effectiveness analysis of CAFE with the aim to balance Europe-wide emission controls against local measures. The model inter-comparison focuses on ambient levels of particulate matter and ozone in urban areas. It addresses health-relevant matrices of exposure (e.g., long-term concentrations) to fine and coarse particles and ozone.
- Projections of future economic activities and their implications on the evolution of energy systems and agricultural activities.

For integrating this variety of information to allow policy-relevant conclusions, CAFE has employed the RAINS model (Amann at al., 2005).

Boundary work at the science–policy interface of the CAFE program

In the field of air quality policy making in Europe a long tradition exists in using scientific information to support negotiations and decisions (e.g., Hordijk 1991; Levy 1993; Castells 1999; Grünfeld 1999; Tuinstra et al. 1999; Bäckstrand 2001; Wettestad 2002; Grennfelt and Hov 2005). Examples of this science-policy interaction are the activities of the United Nations Economic Commission for Europe (UN-ECE) in preparing international conventions and protocols in which countries agree on reducing emissions of atmospheric pollutants (Sliggers and Kakebeeke 2004). Another example is the preparation of EU legislation, e.g. directives on national emissions ceilings and air quality standards (Wettestad 2002). Both examples show an intensive communication process between scientists and policy makers, where knowledge from different scientific disciplines e.g. economy, soil-science, ecology, meteorology and other knowledge sources is integrated in such a way that it provides useful information for decision making. These special communication processes can be referred to as assessment processes (e.g. Farrell et al. 2001). Assessment processes are embedded in a variety of institutional settings, within which scientists, decision makers and other stakeholders communicate to define relevant questions for analysis, mobilise certain kinds of experts and expertise, and interpret findings in particular ways (Farrell et al. 2001). Models, such as IIASA's RAINS model, an integrated assessment model at the transnational level, play a central role, in providing scientific advice in the European air quality policy making process, particularly in setting air quality standards.

How scientists and policy makers interact at the interface between science and policy has been studied empirically in terms of the 'boundary work', through which the boundary between science and policy is maintained (e.g., Gieryn 1999; Jasanoff 1990). The main conclusion of these studies is that it is impossible to find stable criteria that absolutely distinguish science from non-science, e.g., politics. Many social scientists who have studied the relationship between the practices of science and decision making have indeed concluded that these two categories of activities cannot be neatly separated (e.g., Jasanoff and Wynne 1998).

As shown by Tuinstra (2006), who studied the interface between science and policy in the preparation of the 2005 Thematic Strategy on Air Pollution (the Clean Air for Europe – CAFE – program), although the European Commission made a clear institutional separation between risk assessment (science) and risk management (policy), in practice the boundaries between the two are continuously moving. For instance, representatives of member states play a role of "experts" in the expert phase and play a role as policy maker once the proposal for legislation has been made. The decision who constitutes an "expert" and who is a "stakeholder", or more generally who is allowed to contribute to the production of knowledge, is problematic and challenged. Industry actors, for instance, were dissatisfied with their labeling as "stakeholders" instead of "experts" in the CAFE process. Thus the desired separation between risk assessment and risk management was not fully realized.

Tuinstra (submitted) concludes furthermore that the last step in the expert phase, the discussion between the different services of the European Commission, is not transparant at all. It is difficult to trace how the scientific information that was obtained through the CAFE process ultimately influenced the decision making within the Commission.

4. Adaptation and Anticipation in EU and Dutch PM Policy Decisions

4.a. Adaptation

4.a.i. Changes in EU Policy Caused by Planned Adaptation: The 2005 Proposal to Regulate $PM_{2.5}$

At this very moment, negotations between the European Parliament and the Council of Ministers are taking place on the Thematic Strategy on Air Pollution, which will introduce a new $PM_{2.5}$ target (see box 2 for some information on the EU decision-making process). The first daughter directive of 1999, which introduced regulation for PM_{10}, contained a review clause: at a scheduled time (according to the directive, in 2003) the evidence for the health effects of $PM_{2.5}$ should be reconsidered. This review was effectively incorporated in the CAFE program as part of the EU process for reaching an overall Thematic Strategy on Air Pollution, which was scheduled to be finalized in 2004, but was delayed by another year. Even though there was a two-year delay, the 2005 proposal to indeed regulate $PM_{2.5}$ can be classified as an instance of 'planned adaptation'.

Already in 1998 the Commission had published a discussion paper that looked ahead towards an overall clean air strategy that would include different elements of EU air pollution policy and as such enhance the development of cost-effective solutions. This strategy should be renewed in a five-year policy cycle. This 1998 paper was the starting point for the Clean Air for Europe (CAFE) programme. As said, the first integrated clean air strategy was planned to be adopted in 2004. This process was delayed and the Commission presented the Thematic Strategy on Air Pollution in September 2005. The Commission claims that the proposed measures would greatly reduce air pollution while providing benefits to health that would be many times larger than the abatement costs.

In the CAFE programme, six alternative $PM_{2.5}$ policies were considered (the Commission ultimately chose a combination of 2 and 4):

1. Introduce a legally binding requirement to reduce annual average concentrations of $PM_{2.5}$ throughout the territories of the Member States by a given percentage in 2020 relative to the position in 2010 as determined by three years of monitoring of $PM_{2.5}$ concentrations in urban background locations;
2. Introduce a target to reduce annual average concentrations of $PM_{2.5}$ throughout the territories of the Member States by a given percentage in 2020 relative to the position in 2010 as determined by three years of monitoring of $PM_{2.5}$ concentrations in urban background locations;

3. Replace the indicative limit values for PM_{10} for the year 2010 by a legally binding limit value for annual average concentrations of $PM_{2.5}$ to be attained by 2010. Such a limit value would be designed to offer a high degree of protection to the population and would apply everywhere in the territory of the Member States;
4. Replace the indicative limit values for PM_{10} for the year 2010 by a legally binding "cap" for annual average concentrations of $PM_{2.5}$ to be attained by 2010. Such a "cap" or ceiling would be designed to limit unduly high risks to the population and would apply everywhere in the territory of the Member States;
5. Replace the indicative limit values for PM_{10} for the year 2010 by a non-binding target for the annual average concentrations of $PM_{2.5}$ to be attained as far as possible by 2010. Such a target value would be numerically identical to the limit value in option (2) above;
6. Do nothing, i.e. do not introduce any requirement to reduce human exposure to $PM_{2.5}$.

Box 2 The development of a directive or regulation in the EU under the co-decision procedure.

In the co-decision procedure under which environmental legislation in the EU resides, the European Commission has the right of initiative for new legislation proposals. In the expert phase the European Commission (in this case DG Environment) collects technical and other information needed to develop the proposal. A draft policy proposal is subsequently discussed with other Directorates (services) of the European Commission (the so-called inter-services consultation) before being published as an official proposal of the European Commission. In the next phase, the negotiation phase, the European Parliament (representing the European citizens) discusses the proposal and sends its opinion to the Commission. The Commission then sends the amended proposal to the Council of the European Union. The Council of the European Union represents the governments of the EU member states in different configurations. For example in the case of an environmental proposal, the Environment Council, existing of all Environmental Ministers of the EU Member States will discuss the proposal but the Transport and Energy Council will do so as well. If the Council agrees to the amended proposal the legislation can be adopted and the next phase, the implementation phase follows. Otherwise, the proposal will go back to Parliament and Commission for new amendments.

It should be noted that the development of the Thematic Strategy on Air Pollution follows a slightly different procedure. The strategy will not be negotiated in Council and Parliament. The Council will formulate Council conclusions and the Parliament adopt a resolution. Resulting legislative acts like a new air qualitative directive will be decided upon following the normal co-decision procedure.
Source: European Communities, 2003.

For PM_{10}, the Commission's proposal is not to tighthen the PM_{10} standards (reversing an earlier decision in 1999), but instead to introduce a new $PM_{2.5}$ standard. An annual concentration cap is proposed for the finer fraction of particulate matter ($PM_{2.5}$) of 25 μg/m^3 averaged per year that has to be attained by 2010 throughout the entire territory of each member state. After 2010, a reduction of the average urban background concentration of $PM_{2.5}$ is required over the period between 2010 and 2020. The proposal also includes a possibility for derogation of the limit values for particulate matter (PM_{10} and $PM_{2.5}$) by a maximum of 5 years beyond the attainment date if certain criteria are met. Any request for time extension should be accompanied by a plan to ensure compliance within the extended time period.

The European Parliament environment committee voted overwhelmingly in favor of tightened limit values on air pollution in a vote on 21 June 2006. Specifically, MEPs called for more ambitious $PM_{2.5}$ targets. On 26 September 2006, the European Parliament voted to reduce the annual PM_{10} norm to 33 μg/m^3 by the year 2010 and to introduce an indicative $PM_{2.5}$ norm of 20 μg/m^3 in 2010. The PM_{10} norm is stricter than the 40 μg/m^3 originally proposed by the Commission (equal to the current limit value). At the insistence of German conservative MEPs in the environment committee, some flexibility was granted as to how member states should meet the new pollution limits. This flexibility was taken over by the Parliament on 26 September 2006. The Environment Council of Ministers is expected to vote on the proposal in this first reading before the end of 2006.

The change of the original proposal by the Commission to introduce a binding $PM_{2.5}$ target and instead propose a non-binding target was explained by the European Parliament to be due to the uncertainties that still surround the whole PM debate. Therefore, the Parliament proposed a scheduled review, so that the target could become binding if the scientific evidence became stronger.

4.a.ii. Changes in EU and Dutch Policy Caused by Other Processes: The 1999 Decision to Regulate PM_{10} and the Obstacles Met in the Dutch Implementation

The pre-1996 Directives were based on the best scientific evidence available at that time, and in particular the work of the World Health Organisation (WHO), but there had been further research on the effects of air pollution on both human health and the environment, which it was felt should be taken into account. In addition, implementation of the existing Directives revealed a number of problems. It was therefore decided that the European Union should bring air quality limit directives up to date. This led to the 1996 framework directive. In 1999, the air quality standards for 'suspended particles' were replaced by standards for only the smaller fraction of PM_{10}. In the 1980 directive that regulated suspended particles, no review was scheduled. Therefore the 1999 decision to regulate PM_{10} can be classified as an 'unplanned adaptation'.

Besides taking measures at the international level, individual countries and some lower-level governmental actors craft policies that aim to assist in meeting the targets set. Depending on how the Directive is implemented in national law, subnational governmental actors can even have an obligation to create their own policies. Such is the case in the Netherlands. The concentration levels of substances from the first daughter directive (among which PM_{10}) have been an important element in the definition of the zones and agglomerations in the Netherlands (Van Breugel and Buijsman, 2001). The result has been a subdivision of the Netherlands into three zones and six agglomerations. The agglomerations are urban areas with at least 250,000 residents. Moreover, the first daughter directive stipulates the numbers of monitoring stations in the zones and agglomerations, which are in turn dependent on the numbers of residents and the concentration levels. The Directive also contains regulations concerning the monitoring apparatus to be used.

The Dutch government opted for a strict – one could even say 'formalistic' – interpretation of the directive: 'the limit values, set by the European Directive, are interpreted as absolute limits, to be taken into consideration by all authorities at all levels of government in the exercise of all their functions which could have an impact on air quality' (Fleurke and Koeman 2005, 382). The Directive stipulates two limit values for PM_{10}. There is a limit value for the annual average concentration of particulate matter that is primarily intended to offer protection against the long-term effects of particulate matter. This limit value is 40 $\mu g/m^3$. The second limit value concerns the 24-hour concentration of particulate matter. This is primarily intended to provide protection against the short-term effects. Specifically, the Directive stipulates that the limit value for the 24-hour average (50 $\mu g/m^3$) cannot be exceeded for more than 35 days during each calendar year.

Since, like several other European countries, the Netherlands at many locations continuously exceeds the limit values of PM_{10}, non-governmental actors have increasingly challenged local decisions, even those not directly but only indirectly affecting future air quality. As was reported by Koelemeijer et al. (2005), more than 40 spatial plans proposed by Dutch authorities have been appealed by stakeholders since 2001 on grounds of possible breaching of air quality limit values. In about one-third of the cases, the appeal was sustained by the highest court of justice on these matters because of air quality reasons, which resulted in the rejection of the plans. This concerned zoning plans for development of homes and business parks, permits for industrial activities, and plans for road construction or modification. The Dutch jurisprudence has clearly demonstrated the necessity of very detailed impact assessments, before permits for spatial developments can be granted. Moreover, air quality that does not meet the limit values can mean calling a halt to spatial developments, thereby conflicting with the general approach which Dutch authorities and companies have used in the past to assess the consequences of their (spatial) plans.

Koelemeijer et al. (2005) also found in their investigation of other EU countries that only few court cases of this type have occurred elsewhere, while breaches of air quality limit values and the limit values enhanced by the margin of tolerance do occur in other EU countries as well. To explain this situation, they have studied the transposition and application of the first Daughter Directive on Air Quality in seven member states: Austria, Belgium (Flanders), France, Germany, the Netherlands, Sweden and the UK. The Netherlands was found to have implemented the first EU Daughter Directive on Air Quality in a relatively strict fashion compared to other countries. While most countries show exceedances of limit values, and all countries base their policy on the same EU air quality directives, considerable differences exist between member states with respect to the role limit values play in granting permits for new (spatial) developments. This is related to four aspects of the Dutch implementation.

First, the Netherlands enforced a strict legal coupling between air quality policy and spatial planning policy. Not only are all plans that directly affect air quality subject to an impact assessment, but also plans that affect exposure of the population to polluted air. If the impact assessment does not show how all relevant (future) limit values will be met, or how the plan fits into a general policy to meet the limit values, the court may reject it upon appeal. Second, limit values are perceived as absolute limit values, whereas in other countries (Belgium, France, UK), the need to meet a limit value is weighted with other interests when deciding on whether to grant permits. Although Germany also perceives limit values as being absolute, the consequences for granting permits for spatial developments are not as far-reaching as in the Netherlands. Third, limit values apply to the whole of the Netherlands. In principle, these limit values apply in all other countries to anywhere in the outdoors. However, Germany and Austria, at least, infer from the nature of the limit values that they only have to be met at locations where people can be expected to be exposed for a period that is significant compared to the averaging period of the limit value. Finally, in the Netherlands, air quality is assessed with much detail, because it is based on a combination of measuring and modelling with high spatial resolution. Consequently, many places are designated as locations where air quality limit values are breached, particularly in built-up areas close to busy roads. A limited number of countries also employ models with high spatial detail, but many countries only use measurements to assess air quality.

Faced with the severe economic and social consequences of its environmental rules, the Netherlands is now slowly but gradually moving towards a more flexible interpretation of the EU Directive. A more flexible interpretation of the Directive – called the 'moderate' approach by Fleurke and Koeman (2005, 382) – is to argue that 'the limit values should be achieved, in the first place, at national level, through national measures rejecting the interpretation that all activities which have only a limited effect on air quality must be subject to an impact assessment'. Although the limit values are absolute, this does not mean that the Air Quality Order should become the overriding concern in all decisions on spatial planning or infrastructure projects. The approach should be to deal with the large polluters first. The Dutch government took a first step towards this more flexible interpretation by issuing a new Air Quality Order in 2005. Some flexibility has been built in. For example, the European legislation offers possibilities to subtract particulate matter originating from 'natural phenomena' from the measured particulate matter concentrations under certain conditions. Thus, the Netherlands is now subtracting the seasalt contribution to PM_{10}. Furthermore, a possibility to average the air quality over larger areas (compensation scheme) was introduced (although it is not yet clear whether the European Commission will find this acceptable). Finally, the stand-still principle that was enshrined in the old Order was removed, which allows increases in PM concentrations in areas that are still under the limit. However, as Fleurke and Koeman (2005, 383) conclude, 'it is doubtful if this will do the trick as all administrative decision by all authorities will still have to be assessed as to their effects on air quality'.

We can conclude that in the implementation phase national policies adapt to changing scientific insights and societal or political opposition. The changes in the Dutch implementation of the 1999 daughter directive must be characterized as ad hoc. The failure of anticipation of the societal and economic obstacles faced in the implementation of the EU daughter directive is further dealt with in section 4.b.iii.

4.a.iii. Notable Delays in Change and Their Causes

As compared with the United States, it looks as if the introduction of $PM_{2.5}$ norms has been significantly delayed. In 2006, the Europeans are still debating what kind of norms to implement for 2010 that are less strict than the norms that have already been in place in the US since 1997. Again, as was the case for the introduction of the PM10 norms in the EU (the decision was taken in 1996, nine years after the US introduced its PM_{10} norms), there seems to be a delay of nine years. From the US perspective, this may indeed be called a 'delay'. An alternative reading, however, is that US regulators have taken a more precautionary approach, with the facts about $PM_{2.5}$ and its effects on health still being uncertain (see section 4.b.ii). Faced with the same uncertainties, US regulators deemed it necessary in 1997 to regulate $PM_{2.5}$, while EU decision makers found the evidence base too uncertain to warrant immediate measures. But worries were raised in the EU too. Therefore the monitoring of $PM_{2.5}$ has already become mandatory at a small scale since the 1999 directive and a review of the evidence was foreseen within several years after the introduction of the directive. And at the present moment EU regulators do find the evidence strong enough to introduce a norm – though the European Parliament still doubts whether the evidence is strong enough to warrant the introduction of a binding target. It remains to be analyzed what exactly were the societal pressures in the US and EU pro and against the $PM_{2.5}$ norms and how these pressures influenced the final decision making.

4.b. Anticipation

4.b.i. Level of Transparency of Modeling Processes and the Creation of Credibility

In the 1980s and 1990s, scientific and policy communities in longe-range transboundary air pollution have approached each other and developed a mutual dependency: the increased availability of information through the use of models and the trust given to the experts greatly facilitated institutional innovation. Based on a debriefing exercise concerning the use of the RAINS model in international negotiations, Mermet and Hordijk (1989) presented a framework that correlates the role of assessment models to different kinds of policy contexts in which they are used and to the level of use (table 1). The point where parties accept the model as a shared frame of reference for policy making is not easily achieved and requires that the research community negotiates credibility for the science and actively builds trust relationships with the actors involved. In the end, the RAINS model played a major role in the international acid deposition negotiations in the framework of the United Nations Convention on Long-Range Transboundary Air Pollution and became an annex to the United Nations SO_2 protocol. (Hordijk 1991).

Role of model	*User level*	
	Individual	*Collective (joint use by all parties)*
Model as motor of the process	A party promotes the model as an active basis for its position	All parties agree to use the model as reference framework for the process
Model as a source of information	A party uses the model to complement the argumentation of its position	The model is considered by all parties as one source of information used in the process
Model indifferent because marginal or useless	A party is reluctant to move from the political to a more technical ground	The negotiation is so adversarial that "rational" analysis of the problem plays little role
Model undesirable	A party disagrees on the science or fights the model as a tactic in the negotiations politics	Prospects for the use of the models are terrible

Table 1 Types and levels of use of assessment models in the negotiation process (Mermet and Hordijk 1989).

The science–policy interface in the CAFE program was explicitly organized as a much more open and transparant process as compared with the preparation of earlier directives. This was part of an overall strategy in the EU to increase transparency. The CAFE secretariat took great effort to make sure that the work would have credibility, legitimacy and relevance (Tuinstra submitted).

Steps to enhance credibility included e.g. the a special review procedure of the RAINS model. Furthermore bilateral consulations took place with individual countries on inputs for the models used (see Tuinstra, submitted). This enhanced both the credibility and legitimacy of the use of the model for several countries.

4.b.ii. Assessment of Knowledge and Uncertainties in EU PM Policy Making

In this section, we will analyze what uncertainties are present in the PM and health problem and how they are dealt with in European integrated assessment efforts. Even though the evidence from epidemiological studies accumulates and consistently shows statistically significant associations between health effects and PM_{10} or $PM_{2.5}$ concentrations (Pope and Dockery 2006), large uncertainties and controversy remain about the sources, exposure and causes of health effects (RIVM 2002; MNP 2005; Moolgavkar 2005; Maas 2006). On the basis of an expert meeting with Dutch experts on particulate matter and health organised by MNP and Utrecht University in May 2005 (Kloprogge and van der Sluijs 2006), combined with the Impact Assessment of the EU's Thematic Strategy on Air Pollution (COM(2005)446 and 447), we have identified the following key sources of uncertainty in the integrated assessment of the PM and health problem:

a) attribution of effects to individual species of particle (causal fraction) or other pollutants or stressors;
b) quantification of the mortality impact of exposure to fine particles;
c) distribution of risk over subgroups of the population (to what extent is the relative risk age-dependent?);
d) valuation of mortality impacts from particles and other pollutants;
e) assessment of effects of chronic exposure to particles on the prevalence of bronchitis;
f) inter-annual variability in meteorology;
g) uncertainty in cost estimates of measures;
h) emission data;
i) poor understanding of secondary organic particles; and
j) measurement uncertainty.

Below, we use the structure of the *MNP Guidance for Uncertainty Assessment and Communication* (MNP/UU 2003), shown in figure 1, for systematically reflecting on issues of uncertainty management and communication in the case of health risks from PM.

Problem framing. Four problem views can be distinguished in the policy debate on particulate matter: "$PM_{2.5}$ is the problem", "PM_{10} is the problem", "Specific traffic related species are problem (e.g., diesel soot)", and "It is mainly a socio-economic problem (PM not main cause)" (Maas 2006). The conclusions of scientific assessments of the PM problem are critically sensitive to the problem frame chosen, while the present state of knowledge is inconclusive regarding which framing is most adequate. For instance, strong associations can also be found between cardiopulmonary diseases and

Foci	Key issues
Problem framing	Other problem views; interwovenness with other problems; system boundaries; role of results in policy process; relation to previous assessments
Involvement of stakeholders	Identifying stakeholders; their views and roles; controversies; mode of involvement
Selection of indicators	Adequate backing for selection; alternative indicators; support for selection in science, society, and politics
Appraisal of knowledge base	Quality required; bottlenecks in available knowledge and methods; impact of bottlenecks on quality of results
Mapping and assessing relevant uncertainties	Identification and prioritisation of key uncertainties; choice of methods to assess these; assessing robustness of conclusions
Reporting uncertainty information	Context of reporting; robustness and clarity of main messages; policy implications of uncertainty; balanced and consistent representation in progressive disclosure of uncertainty information; traceability and adequate backing

Figure 1

Foci and key issues in uncertainty assessment and communication (MNP/UU 2003).

for traffic noise (Kempen et al. 2002), the quality of housing and the diet of low income families (Eschenroeder and Norris 2003). Even though recent attempts to correct for such confounding effects have strengthened the evidence for low-dose PM effects on health, it can still not be ruled out that the observed health effects are largely caused by an accumulation of other causes in low income neighbourhoods close to highways. The degree to which such interwovenness with other problems is taken into account and the choices made for the system boundary may influence the conclusions. This requires systematic reflection by science advisers. The way uncertainties about the health risks of PM should be dealt with in policy advice depends on the role of such advice in the policy process. In some countries, such as the United States, $PM_{2.5}$ has already been regulated since 1997, while in the European Union such regulation is still being discussed. In the first case, the focus of assessments is more on the effectiveness of existing or proposed regulation than on the need for setting new air quality standards. And thus the types of uncertainties that are most important to deal with differ among these cases.

Involvement of stakeholders. Participation of stakeholders in knowledge production can help to increase the quality of the risk assessment. Participation stimulates the inclusion of more viewpoints, which in turn helps to rule out that important dimensions of the problem are overlooked. Further, participation opens opportunities to make use of local knowledge. The lack of inclusion of such local knowledge frequently forms a barrier to the acceptance of scientific assessments as a shared basis for decision making. Examples of relevant local knowledge on PM include reflections on the (in)adequacy of existing emission monitoring systems and substantive knowledge on the values of parameters in local air pollution models (Yearley 1999). However, in current practice this reservoir of local knowledge is hardly utilised, which is an omission. Participation of stakeholders in assessment can also improve the use of assessments. For instance, in the US, proposals for new air quality standards, such as the revision of the NAAQS for PM proposed in 2005, undergo a public review that aims to build a widely shared scientific basis. As another example: in the Clean Air for Europe programme, over one hundred stakeholder meetings were organised to disseminate results, to share experiences on the use of different policy instruments (including economic instruments), to discuss issues relating to the implementation of current air quality legislation, and to review the uncertainties and their implications.

Selection of indicators. The choice of using particle size to assess and regulate health risks of PM is problematic. Generally speaking, particle size is an imperfect proxy for toxicity. The chemical composition and reactive surface of the particles may be of much more importance, but are difficult to measure and monitor. PM_{10} and $PM_{2.5}$ may not be the most relevant indicators for the health risks from PM; depending on the problem frame chosen, other indicators become more relevant (e.g., specific chemical fractions). If specific chemical PM fractions are suspected to be primarily responsible for the health impacts (e.g., particles emitted from cars), then reducing SO_2 emissions from electric utilities is not an effective way to reduce health risks, despite the fact that $PM_{2.5}$ concentrations (secondary particles) decrease. Furthermore, the precise formulation of indicators used in the regulation of PM varies between countries. For instance, in the United States the 24-hour standard of PM_{10} of 150 $\mu g/m^3$ may be exceeded once a year, while in the EU a standard of 50 $\mu g/m^3$ may not be exceeded more than 35 days per year, which makes comparison between these standards difficult, though not impossible. In this paper, it was concluded from a comparison of both standards using data from 18 monitoring stations in the Netherlands that the EU norm is more strict in most cases. In contrast, it was also observed that the current $PM_{2.5}$ standard in the US (effective since 1997) is more strict than the proposed standard in the EU. Thus the choice and precise definition of indicators makes a huge difference in practice.

Appraisal of knowledge base. There is a broad consensus among scientists and policy makers that PM constitutes health risks that need regulation. However, the evidence from toxicological and biological studies is still weak (Moolgavkar 2005). While there are several plausible hypotheses, it is recognised that we are ignorant of the true underlying mechanism that explains the association between PM and health effects. Only a small number of long-term epidemiological cohort studies have been performed, mainly in the United States. It is questionable whether the results are representative for other countries. Furthermore, it is difficult to determine the exposure to PM (exposure depends on the behavior of individuals, for which assumptions have to be made), to establish a reliable exposure–effect relationship, and to account for multicausality and synergies. Finally, there are bottlenecks in determining PM emissions and concentrations: measurements are often unreliable or not representative for larger areas; and models often give estimates that are not in agreement with measurements.

Mapping and assessing relevant uncertainties. In table 2, the key sources of uncertainty a) through j) are mapped on the uncertainty typology of the Guidance. In thetable we can see that model structure uncertainty and data uncertainty are particularly pertinent in this case, that the quality of the evidence for the causal models is considered problematic, that model assumptions may be subject to subjective choices (cf. Kloprogge et al. 2006) and that the data uncertainty for emissions, meteorology and concentrations are largely characterized by variability and can thus not be fully reduced. This analysis shows that the classical statistical uncertainty methods are not sufficient to deal with the key uncertainties in the PM and health case, and that other methods are implied. Scenario uncertainties can be addressed by scenario analysis techniques. For an assessment of the qualification of the knowledge base for a particular model structure, for example, pedigree analysis (van der Sluijs et al. 2005; Refsgaard et al. 2006) or a model quality checklist (Risbey et al. 2005) can be used. The value-ladenness of a model can be assessed, for instance, by way of critical analysis of assumptions (Kloprogge et al. 2005) or perspective-based scenarios (van Asselt 2000). In CAFE, some of the uncertainties have been analysed by way of sensitivity analysis, focussing particularly on uncertainties in energy demand and agricultural production, emission data and emissions abatement factors, the various ambition levels, or target-setting methods.

Reporting uncertainty information. The uncertainty aspects of the integrated assessment of the PM and health problem have been described in the Impact Assessment. Uncertainty ranges were published in the Impact Assessment for the benefits of the Commission's Thematic Strategy, with as one of the main aims a reduction by 20% of the average urban background concentration of $PM_{2.5}$ in the period between 2010 and 2020. The estimates of total benefits of the Thematic Strategy vary between €37 billion and €119 billion per annum in 2020. These are between seven and 24 times higher than the estimated costs of between €5 and €8 billion per annum.

Box 3 Uncertainty Typology

In order to facilitate communication about the different types of uncertainty that arise in scientific assessments, an uncertainty typology is part of the *MNP Guidance for Uncertainty Assessment and Communication*. The typology is based on a conceptual framework that resulted from a process involving an international group of uncertainty experts most of whom participated in developing or reviewing the Guidance. Uncertainty can be classified along the following dimensions: its 'location' (where it occurs), its 'level' (whether it can best be characterised as statistical uncertainty, scenario uncertainty or recognised ignorance) and its 'nature' (whether uncertainty primarily stems from knowledge imperfection or is a direct consequence of inherent variability). In addition, the typology distinguishes the dimensions 'qualification of knowledge base' (what are weak and strong parts in the assessment) and 'value-ladenness of choices' (what biases may shape the assessment). The typology is presented as a matrix. This uncertainty matrix is used as an instrument for generating an overview of where one expects the most important (policy-relevant) uncertainties to be located (the first dimension), and how these can be further characterised in terms of the other uncertainty dimensions mentioned. The matrix can be used as a scanning tool to identify areas where a more elaborate uncertainty assessment is required. The different cells in the matrix are linked to available uncertainty assessment tools suitable for tackling that particular uncertainty type. These tools are described in a Tool Catalogue that aims to assist the analyst in choosing appropriate methods.

Location of uncertainty ↓		Level of uncertainty *(from determinism, through probability and possibility, to ignorance)*			Nature of uncertainty		Qualification of knowledge base			Value-ladenness of choices		
		Statistical uncertainty	Scenario uncertainty	Recognised ignorance	Epis-temic	Varia-bility	–	0	+	–	0	+
Context												
Expert judgment			d, g			d, g		d, g			g	d
MODEL	Structure		b, e	a, c, e, i	a, b, c, e, i	c	a, c, e, i	b,		C	b, e	a, i
	Implemen-tation											
	Param-eters		b									
	Inputs	f	f, g		g	f		g	f	F	g	
Data		f, h, j	f		h, j	f, h, j		h	j	h, j		
Outputs												

Table 1

Uncertainty Matrix. The labels refer to the sources of uncertainty mentioned in the text. The function of this matrix is to identify the most salient uncertainty types that should be addressed in uncertainty assessment and communication. The typification has been done by the authors.

If we take all the uncertainties into consideration, and add the possibility that regulating size fractions of PM may not be an effective way to increase health, we must admit that not all knowledge that is available makes explicitly part of the decision making process. Although we would not want to defend the claim that knowledge is being actively suppressed in the European context, we do think that other views on the PM–health problem deserve more policy attention.

It is possible to identify in the literature four 'stylized' ways to define the PM problem (see figure 2):
1) '$PM_{2.5}$' – focus on transboundary air pollution and secondary inorganic particles
2) 'PM_{10}' – a large part will be $PM_{2.5}$ but also the abatement of primary emissions of coarse particles becomes part of the strategy
3) 'PM_x' – focus on the traffic related carbonaceous particles & ultrafines, since these are considered to be particularly toxic
4) 'No PM' – focus on living conditions in low income neighbourhoods

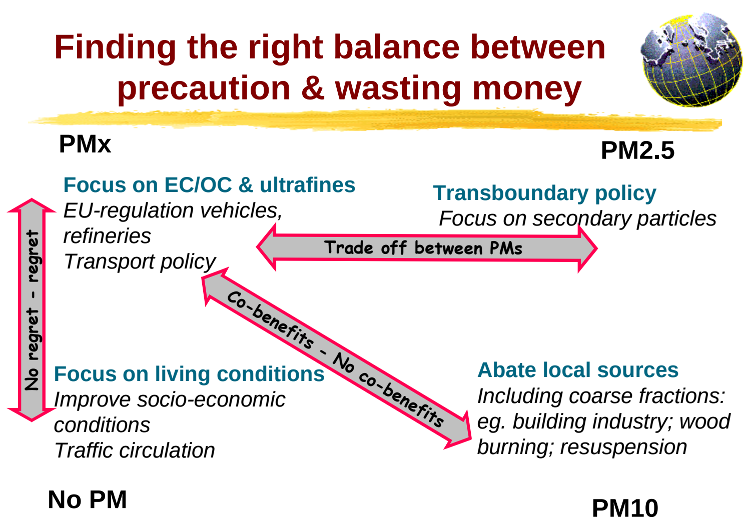

Figure 2
Finding the right balance between precaution and wasting money

We may therefore seriously question whether a sole focus on regulating a certain size fraction of PM (be it PM_{10} or $PM_{2.5}$) is the best policy option to available to improve human health.

4.b.iii. Misanticipation in Dutch PM Policy Making

The case of building projects in the Netherlands provides a clear example of misanticipation. The Netherlands chose a more strict interpretation of the EU air quality limit values than did many other EU countries. The Netherlands also required the use of detailed modeling to help predict the impact of construction projects on air quality. Policymakers did not initially consider the high economic costs of this strict interpretation of the EU air quality limit values, and they did not foresee the strict way the judiciary would turn out to actually enforce the air quality order of 2001. They are now reconsidering their method of interpretation of the standards. The misanticipation in this case was the failure to anticipate the high economic and social costs of the policy.

5. Discussion

In this paper, the EU policy making process for setting PM standards and the Dutch policy making process for implementing them have been reviewed. It seems that with respect to $PM_{2.5}$ standard setting, the EU is clearly on a planned adaptation course. Since 1999, the scientific knowledge is scheduled to be reviewed about every five years. The question remains why the US has chosen a more precautionary approach than the EU. Can this be explained by 'pure politics' and is there a disconnect between science and policy in the EU, given that the European Commission has significant freedom to do with the science what it deems right and their proposals subsequently enter a political negotiation phase? Or, alternatively, is there a serious concern about using the wrong indicator ($PM_{2.5}$) for policies that aim to improve health conditions. The US has abandoned its annual PM_{10} norm. Are we certain enough about what causes the statistical relationship between PM and health effects to justify such a policy change?
We cannot be sure that the health effects of $PM_{2.5\text{-}10}$ are small.

The policy adaptation in the CAFE program to make the science–policy interface more transparent, although being part of a wider EU strategy to increase participation in decision making, seems to be driven largely by instrumental reasons. Reasons of increased quality control and making use of a wider range of relevant wisdom and democratization of science in general (see, e.g., NRC 1996) seemed to have played only a minor role. This may explain the frustration of industry that was not granted an expert role in the process. We cannot easily say that the industry lobby has caused the 'delay' in implementing the $PM_{2.5}$ norm.

In the Dutch case of implementing the EU standards for PM_{10} (and other air pollutants), we found that the Dutch implemented the EU directive in a very strict manner. We do not want to claim here that this was an erroneous decision. If you really want to have clean air, you will have to work for it. The only thing that we wanted to point out is that the Dutch government had misanticipated the social and economic consequences and that it was subsequently forced to adapt its policies on an ad hoc basis. The source of the problems is of course nonattainment of the standards. The Netherlands is not the only EU country facing this problem. Many member countries have problems with attaining the PM standards. The newly proposed EU directive that introduces the $PM_{2.5}$ norm also introduces more flexibility and possibilities for delayed attainment dates for the existing PM_{10} norms. In that sense, the new directive also constitutes an unplanned adaptation at the EU level.

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Anticipation and Adaptation in U.S. Air Quality Policy

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6. Introduction

The U.S. study focuses on four cases within federal U.S. air quality policy. The four cases are the National Ambient Air Quality Standards for particulate matter (PM) and sulfur dioxide (SO_2), the Acid Rain Program, and the Clean Air Interstate Rule. The latter is partly a case by itself and is also part of the Acid Rain Program case because it was implemented recently, in 2005, and a complete retrospective analysis is not possible.

In the air quality policy decisions these cases encompass, policymakers have dealt with scientific uncertainty – they have navigated the hazy boundary between science and policy. This study's broad goal is to understand whether the practices at the science-policy interface made the decisions in one case more robust than those in the others. In the cases: Which processes have helped facilitate the assimilation and surfacing of relevant information? Which have reduced the nastiness of fights over policy options and over the science used to justify them? Which have broadened participation in the development of policy and broadened acceptance of decisions? Examples of successful negotiations of the science-policy boundary could inform the design of future policy institutions, especially as the sciences have been playing an increasingly dominant role in the justification of environmental policy.

One conclusion of the study is that the institutions and circumstances in the PM air quality standard (NAAQS) case have been more successful at handling uncertainty and at updating decisions to new information than the other cases. This has allowed the U.S. to lead other countries in its adoption of protective $PM_{2.5}$ air quality standards, which have generated large health benefits. The unanswered question is how did that program achieve this, especially when compared to the other, less successful, U.S. cases that have many similar characteristics?

To answer this question, this study examines the relationships between these four cases. Doing so enables a richer analysis than would be possible for each case alone. Because of its emphasis on connections, this study does not describe every policy decision related to PM or SO_2 made in the U.S. between 1970 and the present. It mentions important decisions to control mobile sources of pollution, but the analysis focuses on *federal* stationary source policy.

The cases suggest that the PM program was more successful at handling uncertainty and at adapting to new information than the other programs because of a confluence of positive factors. Negative factors like industry using the arguments about the validity of science and the court system to delay policy actions occurred in all cases. But, the PM case benefited from each one of the positive influences identified; this was not true for any other case. The chart in Table I summarizes these findings.

Table I. Summary of characteristics, influences, and outcomes for the four U.S. cases.

Basic Characteristics	**PM NAAQS**	**SO_2 NAAQS**	**Acid Rain**	**CAIR**
Congressional Mandate	●	●	●	
Mandated Reviews	●	●	●	
Ambient Standard	●	●		
Direct Regulation			●	●
Positive Influences				
New Knowledge Assessment	●	●	●	
Legitimate	●	●	●	N/A
Credible	●	●	○	N/A
Salient	●	○		N/A
Court Motivation	●			
Extrnl. Incentv. to Surface Info	●			●
Interest Group Politics	●		●	●
Negative Influences				
Courts Cause Delay	●	●	○	○
Science/Models Challenged	●	●	●	●
Interest Group Politics/Politics	●	●	●	●
Outcomes (Positive)				
Early Warning --> Data Collctn				○
Decisions Anticipate Science	●			
Reviews Completed	●	○	○	
Planned Adaptation Occurs	○			
Adaptation Occurs	●		○	○

○	Partial fulfillment
●	Fulfillment
	No Fulfillment

Strong external incentives also motivated policymakers and others to surface information supporting changes in the PM air quality standards. The U.S. Environmental Protection Agency (EPA), Congress, some portions of industry, and academics all wanted the Acid Rain Program and its novel cap-and-trade program to succeed. The $PM_{2.5}$ standards and the science behind them – which the PM reviews generated and assessed – provided key evidence of the achievements of the Acid Rain Program. Public interest groups supported changes in the PM NAAQS independently of external motivations.

The analysis is divided into two sections. Section 2 includes a timeline of major decisions relevant to these four cases and an overview of the important institutions of the science-policy interface for federal air quality policy in the United States. Section 3 is the bulk of the analysis. It analyzes the cases with respect to adaptation, anticipation, and incentives to surface knowledge. The final section concludes.

In section three, particular focus is placed on understanding whether planned adaptation has enabled robust policy decisions: have mandatory, scheduled policy reviews ever lead to a series of policy decisions, each made with explicit acknowledgement of uncertainty, that are consistent with a retrospective analysis of the best available science? Adaptation to new knowledge is needed as an option for dealing with uncertainty because of the inherent limitations of the natural and social sciences to forecast, or anticipate, the outcomes of policy decisions. But, there are many forces working against adaptation. The study also considers whether broad participation in the formulation of science and modeling processes (transparency) lead to fewer disputes over models and to the incorporation of the best available knowledge.

7. Section 2: Policy background

7.a. Timeline of relevant, major decisions

This section presents a high-level overview of the decisions made between 1970 and 2006 that are relevant to the four cases. Figure 1 also shows this timeline.

In 1970 Congress passed the Clean Air Act (CAA) that required the EPA to set air quality standards (NAAQS) for six criteria pollutants[7] and to review these standards on a five-year interval. In 1971 the EPA set the original standards.[8] The EPA began planning reviews of the six NAAQS in 1976 and scheduled the first review of the SO_2 and PM standards to begin, as a joint endeavor, in 1979.[9] In the 1977 CAA Amendments Congress reinforced the importance of the reviews by directing the EPA to finish all the reviews by 1980 and subsequent reviews at five-year intervals after 1980.

The EPA released the final drafts of the criteria document for SO_2 and PM and two separate staff papers for SO_2 and PM in 1982. The EPA did not give notice of proposed rulemaking for the PM decision until March of 1984. Two complications caused this delay: the EPA Administrator resigned in 1983 and a Reagan executive order required the EPA to submit a Regulatory Impact Analysis for the PM decision to the Office of Management and Budget (OMB). The delay between the completion of the NAAQS review documents and the proposed rulemaking provided American Steel and Iron the grounds to bring a lawsuit

[7] The six criteria pollutants are particulate matter, sulfur dioxide, nitrogen oxide, carbon monoxide, ozone, and lead.
[8] The FR citation for the original PM NAAQS decision is (36 *Fed. Reg.* 8186)
[9] 44 *Fed. Reg.* 56731

against the EPA in 1984. American Steel and Iron argued that the proposed rulemaking for PM did not "accurately reflect the latest scientific knowledge" as required by the CAA.[10]

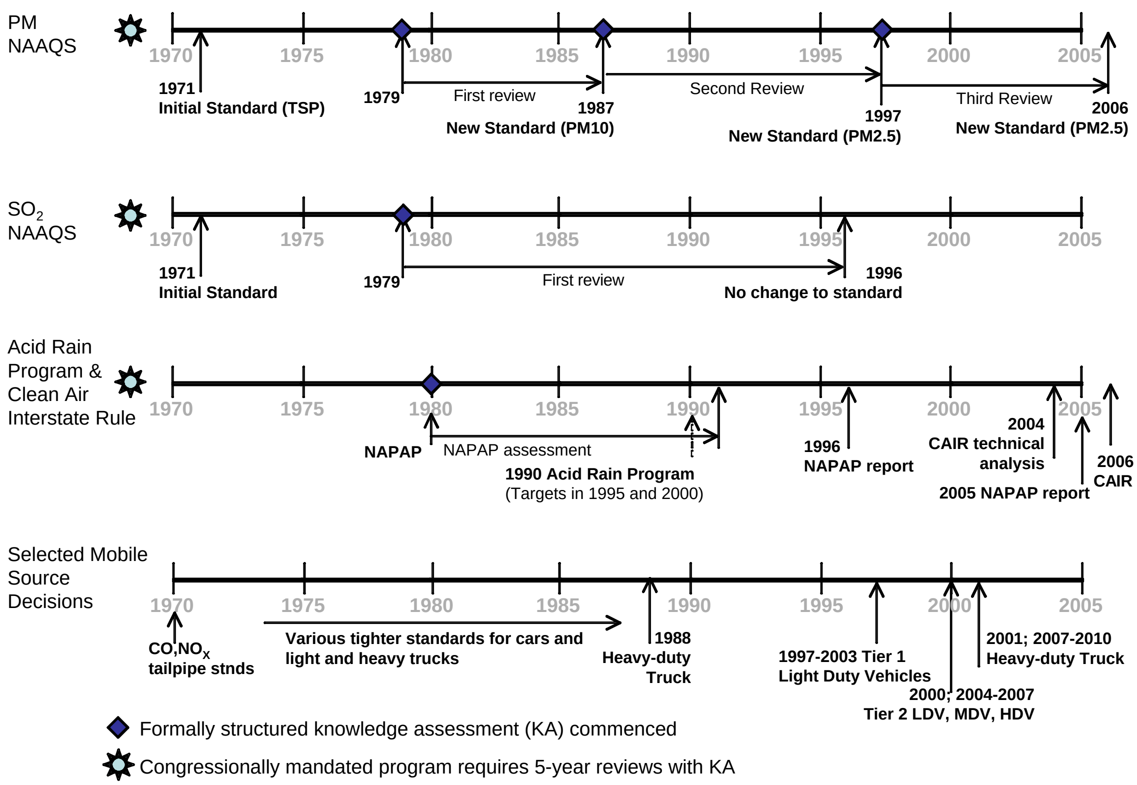

Figure 1. *Timelines of major SO_2 and PM federal policy decisions in the U.S.*

With this lawsuit and the expectation of similar ones as motivation, the EPA decided to write addenda to the original 1982 Criteria Document for SO_2 and PM in 1985. The EPA released these documents in 1986. The EPA Administrator signed the final rule that updated the standard for PM in July of 1987.[11] The change in PM standard dissolved the standard for Total Suspended Particulates (TSP) in favor of a standard for PM with a mean aerodynamic diameter smaller than or equal to 10 microns (PM_{10})[12].

Other policy actions relevant to both SO_2 and PM took place during this period. In 1980 Congress passed the Acid Deposition Act that created the National Acid Precipitation Assessment Program (NAPAP). The motivation for this program was to understand relationships between SO_2 emissions and acid rain and between acid rain and environmental

[10] CAA §108(a)(2)
[11] 52 *Fed. Reg.* 24634
[12] The indicator of PM_{10} actually refers the aerodynamic particle diameter for which the efficiency of collection is 50%. This means that the particle measurement techniques don't exclude larger particles, but collect them with decreasing efficiency. Smaller particles are collected with increasing efficiency, up to 100%. If ambient samplers are used with a cutpoint of PM_{10}, the total amount of particles with diameters less than 10 µm is reliably measured.

damage. NAPAP's focus, unlike the NAAQS process, was not the health impacts of SO_2 and its secondary pollutants, like acid aerosols – a specific type of particulate matter (PM).[13]

The ADA and NAPAP were the foundation for the Acid Rain Program in the 1990 CAA Amendments. In a landmark step, Congress created the first large-scale cap-and-trade program, which regulated SO_2 emissions from stationary sources (Titles IV and V of the 1990 CAA Amendments). Precursors of acid rain are SO_2 emissions and, to some extent, nitrogen oxide (NO_X) emissions.

The Acid Rain Program and its permit-trading program regulate SO_2 but, interestingly, they were not created to help states attain the SO_2 NAAQS. By the 1980s, almost all states were in compliance with the original SO_2 NAAQS. The Acid Rain Program reduced SO_2 emissions directly, but the reductions were not needed to meet the SO_2 NAAQS.[14] The first phase of the Acid Rain Program took effect in 1995, resulting in reductions of about 40% from affected stationary sources (from about 16 million to about 9 million tons).[15]

In 1996 the first review of the air quality standards for SO_2 ended with no change to the standard. This review was the same one that started in 1979 with the review of the PM standard. It was the Acid Rain Program and not the SO_2 air quality standards that drove reductions in SO_2 in the 1990s. Although the Acid Rain Program and the SO_2 NAAQS both focus on SO_2, the two policy mechanisms were not directly related in that decisions for one did not compel decisions for the other.

The EPA commenced its second review of the NAAQS for PM in 1987. The second review ended in 1997 with the creation of a standard for PM with diameter less than 2.5 microns ($PM_{2.5}$) in addition to a tightening of the PM_{10} standard. Also in 1997, the EPA announced the start of the third review for PM. The second phase of the Acid Rain Program took effect in 2000, requiring further reductions of SO_2 emissions from stationary sources. Between 1999 and 2001, the courts vacated the new standard for PM_{10} but the U.S. Supreme Court upheld the standard for $PM_{2.5}$.[16] In 2004, the courts mandated that the third review of the PM NAAQS be completed by 2006.[17] On September 21 of 2006 the EPA announced its new, more stringent standards for $PM_{2.5}$.[18]

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[13] Sulfates are a type of particulate matter.
[14] See, "Sulfur dioxide National and Local Trends in Sulfur Dioxide Levels," at http://www.epa.gov/air/airtrends/sulfur.html last viewed October 2, 2006.
[15] See, Parker, L. "Implementing Acid Rain Legislation," Congressional Research Service Issue Brief for Congress, November 15, 1994, at http://ncseonline.org/NLE/CRSreports/Air/air-8.cfm#Implementation%20--%20SO2 last viewed August 15, 2006 and EPA Clean Air Market's – Acid Rain Program: Overview at http://www.epa.gov/airmarkets/arp/overview.html#phases last viewed August 15, 2006.
[16] American Trucking Associations v. EPA, 175 F. 3d 1027, 1055-56 (D.C. Cir. 1999) and Whitman v. American Trucking Associations, 531 U.S. 457 (2001).
[17] Consent agreement, July 2003, C.A. No. 03-778 (ESH), *American Lung Association, et al v. the U.S. Environmental Protection et al. (EPA)*, U.S. District Court for the District of Columbia and Joint Status Report, September 2004,C.A. No. 03-778 (ESH), *American Lung Association, et al v. the U.S. Environmental Protection et al. (EPA)*, U.S. District Court for the District of Columbia.
[18] See EPA's "Regulatory Actions" Website at http://epa.gov/pm/actions.html last viewed September 23, 2006. Also see 71 *Federal Register* 27, Thursday, February 9, 2006, pg. 6718

Although attainment of the SO_2 standard has not been a challenge for most U.S. states, there are still many counties of the Eastern U.S. (and some areas of California, Montana, Colorado, and Texas) that have not attained the PM standard. Since the NAAQS were established, the state and federal governments have used reductions in both mobile and point source emissions to try to attain the standards. Mobile source regulations include those on light-duty vehicles (LDV), heavy-duty vehicles (HDV), and on nonroad sources including locomotives, marine sources, and equipment like lawnmowers. Stationary source regulations include those on power plants and other large industrial facilities.

There are many examples of federal regulations intended to reduce PM, SO_2, and other emissions from mobile sources. For example, since 1968 the CAA has required the EPA to set standards for the levels of pollutants in the exhaust of gasoline power vehicles (including cars, light-duty trucks, and SUVs). Beginning in 2004, the Tier 2 Tailpipe standards required levels of PM of 0.01 g/mile or lower and also reduced the levels of other pollutants allowed in vehicle exhaust. The EPA also implemented Tier 2 sulfur regulations for reformulated gasoline and diesel in 2004.[19]

In the case of stationary sources, the CAA generally encourages states to develop their own regulations. The CAA gives the EPA the authority to regulate stationary sources when the interstate transport of pollutants and precursors is a problem. The EPA has used this authority recently to help states in the Eastern U.S. achieve the PM and ozone NAAQS by reducing interstate transport of PM, ozone, and their precursors SO_2 and NO_X: in March of 2005, the EPA proposed the Clean Air Interstate Rule (CAIR). In response to eleven petitions for reconsideration from industry, the EPA agreed to reconsider it in November 2005. In March of 2006 the EPA determined that CAIR should remain unchanged.[20]

7.b. Overview of science-policy interfaces

The institutions and processes of the science-policy interface vary in the U.S. between different situations. The NAAQS process has an unusually well defined science-policy interface because each review of the standards requires a formal knowledge assessment and because the Clean Air Science Advisory Committee (CASAC) is heavily involved. The science-policy interfaces in the cases of the Acid Rain Program and the Clean Air Interstate Rule are different from each other and from that for the NAAQS. Congress created the NAPAP in 1980 to assess the science behind acid rain and its impacts. NAPAP was a funding source for relevant research and an organization to create and collate scientific knowledge and to answer policy questions. It informed the development of the Acid Rain Program, although the knowledge assessment process lacked some important characteristics discussed later. The science-policy interface for the Clean Air Interstate Rule did not include

[19] See Union of Concerned Scientists, "The Plain English Guide to Tailpipe Standards," at http://www.ucsusa.org/clean_vehicles/vehicles_health/the-plain-english-guide-to-tailpipe-standards.html last viewed September 23, 2006.
[20] See U.S. EPA, "Clean Air Interstate Rule: Regulatory Actions," at http://www.epa.gov/cleanairinterstaterule/rule.html last viewed August 15, 2006.

a separate organization that helped the EPA review scientific knowledge prior to the rulemaking.

The details of science-policy interfaces are important for understanding policy outcomes. These interfaces are the avenues through which policymakers become informed about science and through which increasingly large parts of the justifications for decisions are created. This section briefly overviews the science-policy interfaces for the four cases and highlights the differences between them.

7.b.i. The NAAQS science-policy interface (Cases: PM and SO_2 NAAQS)

Title I of the CAA mandates the setting of National Ambient Air Quality Standards (NAAQS) for six criteria pollutants, including SO_2 and PM. The Environmental Protection Agency (EPA) must set primary standards for each pollutant to protect public health.[21] The CAA also mandates a review of each air quality standard every five years. The standards are health-based standards and are ostensibly based on the best-available scientific understanding of the health effects of ambient air conditions.[22] Title I of the CAA recognizes that the proper level for each standard cannot possibly be knowable at a given time due to scientific uncertainty.[23] The reviews mean to ameliorate this problem.

The labor-intensive NAAQS review process involves four offices of the EPA, the Clean Air Science Advisory Committee (CASAC), the public, and other government agencies like the Office of Management and Budget. Section 109 of the 1970 CAA created CASAC specifically for the purpose of aiding the EPA in its periodic review of the NAAQS. The CAA requires that CASAC have seven members and contain one medical doctor, one representative from a State air quality agency, and one person from the National Academies of Science. Additional CASAC review panels are formed for each NAAQS review since the amount of work required of CASAC for each pollutant's review is so large. The CASAC NAAQS panels consist of CASAC proper as well as expert "consultants" from academia and industry. One result of the advisory board requirement is that the EPA must directly involve at least some non-government scientists in the air quality policy process.

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[21] The agency can also set secondary standards to protect "welfare" more generally, such as the environment and livestock.

[22] CAA §109(b)(1) states that the NAAQS must be based on scientific criteria and allow "an adequate margin of safety…requisite to protect the public health." Also CAA §108(a)(2) states: "Air quality criteria for an air pollutant shall accurately reflect the latest scientific knowledge useful in indicating the kind and extent of all identifiable effects on public health or welfare which may be expected from the presence of such pollutant in the ambient air, in varying quantities."

[23] The House Committee on Interstate and Foreign Commerce recognized this challenge explicitly in its report on the 1977 CAAA. It stated that the language in the act was "intended to emphasize the necessarily judgmental element in the task of predicting future health risks of present action and to confer upon the Administrator the requisite authority to exercise such judgment … the committee does not intend this language as a license for 'crystal ball' speculation. The Administrator's judgment must, of course, remain subject to restraints of reasoned decision-making." U.S. Congress, House, Committee on Interstate and Foreign Commerce, *Clean Air Act Amendments of 1977,* House Report No. 95-294, to accompany H.R. 6161 (95th Congress, lst session) (Washington, D.C.: U S Govt Print Office, 1977)

The NAAQS process creates three major documents. These are the Air Quality Criteria Document (CD), the Staff Paper (SP), and the Federal Register publications that propose and justify the initial and final rulemakings. The Air Quality Criteria Document (CD) is a comprehensive science assessment document for which the EPA office The National Center for Environmental Assessment[24] is responsible. The role of the SP is to translate the CD from a document intended for a scientific audience to one for a lay, policy audience (NAAQS PRW 2006). Additionally, the SP includes a quantitative risk assessment performed by EPA's Office of Air Quality Planning and Standards. The EPA publishes the proposed and final rules in the Federal Register and includes their justification for the rulings in those documents. CASAC must approve each of the documents with a closure letter. Section 207 of CAA also requires that the EPA explain any differences between the NAAQS rulemaking and CASAC recommendations.

7.b.ii. Knowledge Assessments

For knowledge assessments to succeed they must tradeoff being tied close to the policy so that they address relevant and timely questions while still being sufficiently distant to avoid being corrupted or coerced. Literature out of Harvard's Global Environmental Assessment Project suggests that, to succeed, knowledge assessments must be credible, salient, and legitimate.[25] They must be credible because they must be scientifically and technically believable. Saliency is the criteria that a knowledge assessment must address questions that are relevant to the policy issue. A knowledge assessment is legitimate if those involved in the policy process consider it to be fair, or conducted in a manner that accounted for their perspective.

The PM NAAQS reviews

The PM NAAQS reviews fill all of these criteria, the reviews have maintained credibility and relevance. The use of non-EPA scientists and the advisory board, CASAC, lend credibility to the creation of the criteria document in the NAAQS review process. The criteria documents collate all the relevant science; they are the bulk of the knowledge assessment for the reviews. The need to update the standard and the dependence of other aspects of the NAAQS review, like the staff paper, on the criteria document help motivate its timeliness and relevance. The staff paper translates the criteria document to a form that is even more relevant for policymakers. A planning process that outlines the chapters of the criteria document and the staff paper also assists. The diverse composition of CASAC and the public comment periods for the criteria document contribute to establishing its legitimacy.[26]

[24] http://cfpub.epa.gov/ncea/

[25] Cash, David, Clark, William C., Alcock, Frank, Dickson, Nancy, Eckley, Noelle and Jäger, Jill, "Salience, Credibility, Legitimacy and Boundaries: Linking Research, Assessment and Decision Making" (November 2002). KSG Working Papers Series RWP02-046.

[26] See McCray 2003 and Powell 1997 for further discussion.

One thing to note, however, is that court orders or Congressional actions motivated the completion of each successful PM review processes. The 1977 CAA Amendments Congress required the EPA to finish the first reviews for all six NAAQS by 1980. This did not actually motivate the completion of any reviews by 1980 but the PM NAAQS was the first that the EPA completed in 1987.[27] In 1994 the Federal District Court of Arizona ordered the EPA to finish the second review of the PM NAAQS by 1997.[28] In a 2004 court case the American Lung Association again demanded that the EPA complete a PM review and the Federal District Court of the District of Columbia ordered the EPA to finish the review by 2006.[29]

The SO_2 NAAQS review

For the SO_2 NAAQS, the court cases brought by interest groups against the EPA did not attempt to motivate new reviews. This helps explain the slowness and lack of reviews for the SO_2 NAAQS. The American Lung Association and the Environmental Defense Fund focused their attention on challenging the EPA's 1996 decision not to change the SO_2 NAAQS. The U.S. Court of Appeals for the D.C. Circuit found, in 1998, that the EPA should further review and explain its decision not to establish a new 5-minute NAAQS for SO_2. In response to that remand, the EPA reviewed the information on SO_2 and decided, in 2001, to continue with their initial decision not to implement a 5-minute SO_2 standard. They did agree to help states decide whether local conditions warranted this type of standard on an individual basis.[30] Thus, while the SO_2 review officially ended in 1996, the EPA's work on it did not end until May of 2001. The EPA announced another review of the SO_2 NAAQS five years later on May 15, 2006.[31]

Comparison of the SO_2 and PM NAAQS knowledge assessments

The assessment processes for the PM and SO_2 NAAQS are the same, at least superficially. But, the characteristics that led to highly credible, salient, legitimate knowledge assessments in the case of PM did not prevent political concerns from influencing the results of the SO_2 review. The EPA commenced a joint NAAQS review for both PM and SO_2 in 1979. By 1982 they had decided to consider the two pollutants separately. The EPA's argument for disconnecting the SO_2 and PM reviews was scientific, though its motivation was arguably political. The EPA's political concerns were that the continued joint assessment of PM and SO_2 would lead to the creation of a strict and costly regulatory program for SO_2 through a

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[27] The EPA should have completed all reviews by 1976 but had been completed none of them by 1977.
[28] American Lung Association versus Browner, DC of AZ, October 6, 1994.
[29] Consent agreement, July 2003, C.A. No. 03-778 (ESH), *American Lung Association, et al v. the U.S. Environmental Protection et al. (EPA)*, U.S. District Court for the District of Columbia and Joint Status Report, September 2004,C.A. No. 03-778 (ESH), *American Lung Association, et al v. the U.S. Environmental Protection et al. (EPA)*, U.S. District Court for the District of Columbia.
[30] EPA, "National Ambient Air Quality Standards for Sulfur Oxides (Sulfur Dioxide); Availability of Information," *Federal Register* **66** (6), January 9, 2001, pg. 1665. (66 FR 1665).
[31] EPA, "Science Assessment for Sulfur Oxides," *Federal Register* **71** (93), May 15, 2006, pg. 28023 (71 FR 28023).

"backdoor" rather than in a direct manner like the later Acid Rain Program.[32] The science at that time, in subsequent PM reviews, and then later as the motivation for the Clean Air Interstate Rule, suggested that types of PM formed by mechanisms involving SO_2 were an important fraction of the PM pollution that endangers human health.

7.b.iii. The Acid Rain Program science-policy interface

In Title VII of the 1980 Acid Precipitation Act (APA), Congress created the National Acid Precipitation Assessment Program (NAPAP) to 1) "identify the causes and sources of acid precipitation" and 2) to "evaluate the environmental, social, and economics effects of acid precipitation.[33] The intent of Congress was for NAPAP to be a policy-science interface. The APA stated the intent of Congress to use the results of the NAPAP research program to "take action to the extent necessary and practicable" to limit the emissions that are the sources of acid rain and to ameliorate acid rain's harmful effects.[34]

While the NAAQS review process has generally succeeded in transferring undistorted knowledge across the science-policy interface, NAPAP did not perform as well. The NAAQS process balances the need for separate, credible scientific knowledge assessment and connected policy advice by formulating both a criteria document and a staff paper with those respective purposes. In advance of April 1990 when Congress devised and passed the Acid Rain Program, NAPAP was not able to balance these tasks.

NAPAP focused on the science side of the science-policy interface. Their 1984 report stated, "Decision makers, not researchers, must decide the level of scientific information necessary for decision making. For example, scientists have the task of relating the response of ecosystems to the amount of acid deposition they receive; but it is the role of the policy maker to determine the acceptable level of response … considering the social costs and benefits in addition to other factors" (NAPAP 1984).

Studies criticize NAPAP because it did not address policy questions.[35] These criticisms focus on NAPAP's failure to analyze alternative policy choices and issues like the relative costs and benefits of different proposals for SO_2 reduction targets prior to Congress' decision in April of 1990. Congress passed the Acid Rain Program without attention to its costs and benefits. This is evident because, based on NAPAP's benefit-cost analysis from September of 1990, the program's benefits would not have outweighed its anticipated costs.[36]

In accordance with its 1984 statement, NAPAP did create and fund new, peer-reviewed and published research about acid rain and its environmental effects. But, even this aspect of NAPAP's was not timely. It published its first major reports in September of 1990 and in 1991 (NAPAP 1990, 1991), while the Senate and House of Representatives passed the

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[32] Powell 1999 pg. 242 reference to endnote 5, pg. 263
[33] 42 USC § 8901
[34] 42 USC § 8901
[35] See Rubin *et al.* 1992 and Herrick 2000
[36] Portney 1990

Clean Air Act Amendments in April of 1990 and the President signed them in November 1990. Although NAPAP did not evaluate alternative policy choices, it did succeed in enabling informal communication between Congress and scientists. This informal conversation may have helped make the implementation of the Acid Rain Program politically possible.[37]

In addition to their hesitancy to consider direct policy issues, NAPAP's work prior to 1990 did not focus on or even anticipate the most import impact of the Acid Rain Program: its health benefits. By 1987, the NAAQS process for PM had recognized the potential health benefits that could come from reductions in SO_2 and the sulfate compounds that are both types of PM and components of acid rain. But, in NAPAP's 1996 report, they highlighted these health benefits in their table on "Selected Policy-Relevant Developments Since 1990" (NAPAP 1998, pg. 44). It is these health benefits – largely ignored by NAPAP in 1990 – that have been the *ex post* justification for the Acid Rain Program. They have dominated the program's estimated benefits.[38]

Congress reauthorized NAPAP in under Title IX of the 1990 CAAA and their subsequent activities have been more policy focused. In 1993 NAPAP adopted a framework, called the Tracking and Analysis Framework, for their assessments of the ongoing costs, benefits, effectiveness, and scientific basis of the Acid Rain Program (NAPAP 1998). NAPAP produced reports in 1998 and in 2005. Although these reports suggest that the benefits of the Acid Rain Program far surpass its costs, NAPAP has still not recommended changes in the targets of the Acid Rain.

7.b.iv. The Clean Air Interstate Rule science-policy interface

CAIR is not a congressionally mandated program and the U.S. legal system requires the EPA to justify their authority to regulate. In this case, the EPA's authority stems from the Clean Air Act. The CAA gives the EPA the authority to regulate the emission of criteria pollutants from stationary sources when interstate transport is a cause of NAAQS non-attainment.[39] The EPA's justification for CAIR has six components, as discussed in the EPA's preamble to the rule.[40] The four of these that span the science-policy interface are:

1) Non-attainment areas for $PM_{2.5}$ exist in the northeastern United States and interstate transport contributes to non-attainment;
2) Analysis indicates that achievable emissions reductions of SO_2, and to some extent nitrogen oxides, will result in the desired air quality improvements;
3) Analysis indicates that the regulation's incentives will have the intended effect on emissions from the targeted sources; and
4) That the benefits of improving air quality will outweigh the costs and that CAIR is a particularly cost-effective way to achieve the emission reductions.

[37] Herrick 2000
[38] See Burtraw *et. al.* 1998 and Chesnut and Mills 2005
[39] The EPA has the authority to regulate under CAA §110(a)(2)(D).
[40] 40 CFR Parts 51, 71, 73, 74, 77, 78, and 96

To maneuver through the science-policy interface for CAIR, the EPA borrowed from the analyses that supported the PM NAAQS and the Acid Rain Program. The EPA did not establish separate entities to review the science behind CAIR. In defense of their overall rational for regulating SO_2 and NO_X emissions as a way to decrease ambient concentrations of $PM_{2.5}$, the EPA references the 2004 PM Criteria Document. They also generally reference a report by NARSTO, a public-private partnership that aims to improve Air Quality Management in North America, called "Particulate Matter Science for Policy Makers – A NARSTO Assessment" (February 2003). EPA performed the modeling and data analysis to support the other aspects of CAIR's justification in house.

8. Section 3: Analysis of timeline of PM policy decisions

This section analyzes the cases with respect to five outcomes related to adaptation.

1. Did the mandated reviews of the programs occur?
2. Did planned adaptation occur? That is, did the reviews lead to a successful reassessment of policy decisions and their justification and did the reviews lead to logically consistent changes in policy?
3. Have subsequent scientific findings reinforced or been consistent with the decisions' justifications?
4. Did the recognition of uncertainties or early warning signs solicit the collection of data for use in future decisions?
5. Did adaptation occur? That is, did changes to the policy occur based on new knowledge but not as a result of scheduled reviews?

Of these five, the third deserves more explanation.

It is a desirable outcome for advances in science, an ever-changing process, to reinforce the original basis for policy decisions. Policy decisions are made under uncertainty. If policy decisions are consistent with new science, then the decisions have correctly anticipated the trends in science. Success in this area does not require perfection, especially if adaptation is employed as a tool to deal with uncertainty. If each policy decision in a series of adaptive decisions is made based on the best available information, then as more uncertainties are resolved, both science and policy should converge toward an acceptable outcome. If policy decisions are not based on or adapted to the best available information then it is more likely that future understanding will show that the justifications for those policies were not correct and the outcomes far from ideal.

8.a. Planned adaptation and successful reviews

Within these cases, the search for instances of planned adaptation used as a means to deal with uncertainty yields only one successful example: the PM NAAQS. This is true despite the existence of congressionally mandated scientific review requirements in all three major cases. The requirements for the PM and SO_2 NAAQS are identical: the CAA requires five-year reviews of the air quality standards and their scientific bases. In the case of the Acid Rain Program, Title IX of the CAA Amendments reauthorized NAPAP so that it could review the progress and effectiveness of the Acid Rain Program (including the cap-and-trade program) and any new applicable science.

A cursory glance at the timelines show that the mandated reviews for the PM NAAQS occurred more often (three times) than for the other cases. For the SO_2 NAAQS, the EPA conducted one review. NAPAP produced reports reviewing the Acid Rain Program but these were not intended to alter, or adapt, the program. CAIR is too new to analyze. That multiple, explicit reviews of the PM standards occurred differentiates case, although the reviews occurred more slowly than required and were motivated by court orders on two of three occasions.

The PM NAAQS reviews resulted in changes to the air quality standard and this further distinguishes it. The EPA began a joint review of the SO_2 and PM NAAQS in 1979. While the PM NAAQS review ended in 1987 with an update in the standards, the SO_2 review did not end until 1996 and the EPA made no change to the standard. The SO_2 NAAQS has never been changed. The Acid Rain Program targets have also never been changed. CAIR can be seen as a response to the need to update the Acid Rain Program.

It is not a requirement of successful planned adaptation that policies be changed.[41] In the case of the SO_2 NAAQS, the lack of changes to the standard between 1979 and the present is not consistent with the implementation of the Acid Rain Program or CAIR. These programs require more reductions in SO_2 emissions than needed to attain the SO_2 NAAQS in most states. That another program would be needed to achieve extra emission reductions is inconsistent with the mandate that the SO_2 NAAQS should use the latest science to devise air quality standards to protect the public health and welfare with a margin of safety. Because the SO_2 reductions mandated in CAIR are based on the PM NAAQS, the SO_2 NAAQS results are also inconsistent with those for the PM NAAQS.

The Acid Rain Program is now celebrated as a success because its benefits outweigh its costs. But this success should be considered an accident as the program was not initially designed to achieve these acclaimed benefits nor has the program been updated to reflect them. The Clean Air Interstate Rule, of 2005, is the EPA's response to the disconnect between the politically accepted knowledge tying SO_2 emissions to health effects and the lack of changes to the Acid Rain Program. Although the mechanisms through which SO_2 emissions harm public health are still uncertain, the scientific evidence of a connection is strong enough to have earned the causal chain acceptance in the PM NAAQS assessments

[41] However, it is likely that given new relevant information, one will prefer to change ones strategy. Students of game theory will remember the Monty Hall game.

and as the claimed *ex post* justification for the Acid Rain Program's required SO_2 emission reductions.[42]

8.a.i. Updates to PM standard have been consistent with new science

The changes to the PM standard have been consistent with the progress of science. The knowledge assessments for the PM NAAQS reviews have shown room for further improvement, but they have not shown that the updates to the PM NAAQS were in error. The original indicator for PM was total suspended particulates (TSP). The major decision of the 1989 review was to change the indicator to focus on particles less than 10 microns in diameter (PM_{10}). During the 1979 to 1987 review, the EPA considered science that suggested that a separate standard for particles with diameters less than 2.5 microns ($PM_{2.5}$) was warranted. In the next review, which ended in 1997, the EPA implemented separate standards for PM_{10} and $PM_{2.5}$. Recently, in September of 2006, the EPA made the daily standard for $PM_{2.5}$ more stringent. They also repealed the annual PM_{10} standard but maintained the more stringent daily PM_{10} standard.

The NAAQS reviews and decisions have anticipated the trend in science toward understanding that particles with smaller diameters are more harmful to public health than larger particles. This trend is now appreciated internationally.[43] Particles with diameters between 2.5 and 10 microns do have some impact on health; hence the EPA's decisions to implement a PM_{10} standard in 1987 and to maintain it are not in error.

The EPA, somewhat inconsistently, formally rejected the $PM_{2.5}$ indicator in 1987 for a number of reasons that were still valid in 1997 when the EPA imposed a $PM_{2.5}$ indicator.[44] The agency stressed the uncertainty around the science supporting a $PM_{2.5}$ standard in 1987 (EPA 1982 CD, See Appendix for details). Some of these uncertainties were resolved because of new epidemiological studies by the 1997 review, but some were not. For example, the toxicological mechanisms through which different types of PM impact health are still uncertain (EPA 2004 CD). The EPA justified their $PM_{2.5}$ standard in 1996 largely on new epidemiological studies and on evidence that the two size fractions ($PM_{2.5}$ and $PM_{2.5-10}$) vary

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[42] For a summaries of the benefits of the Acid Rain Program see Burtraw *et. al.* 1998 and Chesnut and Mills 2005. For summaries of the uncertainties surrounding the connections between SO_2 and PM and health effects see the EPA's Criteria Documents (EPA 1982 CD, 1996 CD, 2004 CD).

[43] See for example, World Health Organization, "WHO air quality guidelines global update 2005" *Report on a working group meeting*, Bonn, Germany, 18-20 October 2005.

[44] The EPA's logic in the 1987 Federal Register publication justifying their new PM_{10} standard stated, first, that the EPA rejected the $PM_{2.5}$ standard because the respiratory tract sees the deposition of a mixture of $PM_{2.5}$ and PM_{10} in both the tracheobronchial and alveolar regions: "The mixing of these size fractions in the respiratory tract and the heterogeneity within each fraction therefore blurs the distinction between the fractions in terms of health effects".[44] Second, PM_{10} is associated with various health problems: they believed that not all health effects come from $PM_{2.5}$ alone. Also, a $PM_{2.5}$ was standard not warranted in addition to the PM_{10} standard because 1) fine PM is 40 to 70% of PM_{10} so PM_{10} standard does cover fine PM to some extent, 2) epidemiological studies do not separate the effects of $PM_{2.5}$ and PM_{10} fractions so there is not a basis for a $PM_{2.5}$ standard; and 3) it may be more "appropriate to consider the addition of chemical-specific (e.g., acid aerosols) standards rather than a fine [$PM_{2.5}$] particle standard in future primary standard revision". From EPA's final decision on the PM NAAQS in 1987, as published in the Federal Register (52 FR 24634, July 1, 1987)

separately in time and space. Some of this evidence was already available in the earlier review (EPA 1982 CD).

It is possible that the EPA hesitated to introduce a $PM_{2.5}$ standard in 1987 because it would have required more costly emission reductions from industry than the PM_{10} standard.[45] The uncertainty around $PM_{2.5}$ was more important as industry would be more motivated to contest a $PM_{2.5}$ standard. The SO_2 abatement that took place under the Acid Rain Program between 1990 and 1995 (which reduced $PM_{2.5}$ levels) and new studies implicating $PM_{2.5}$ then allowed the EPA to impose a $PM_{2.5}$ standard in 1996.

The supposition that the EPA understood that industry would have fought fiercely against the $PM_{2.5}$ standards in 1987 is supported by the attack on the $PM_{2.5}$ standards that occurred starting in 1996 when the EPA first proposed a $PM_{2.5}$ standard. Industry tried to discredit the new science that supported the $PM_{2.5}$ standards in 1996 – in particular, the Harvard Six Cities Study.[46] They also sponsored a public campaign that, among other things, created television ads claiming that the EPA's rules were so strict that they would prohibit barbecues and fireworks.[47]

By 1996, the evidence that $PM_{2.5}$ was a problem supported the EPA's $PM_{2.5}$ standards despite industry's fierce opposition. The controversial Six Cities Study and the American Cancer Society study that linked small particle (e.g. $PM_{2.5}$) pollution to cardiovascular disease were particularly important.[48] The Six Cities Study was later validated and the EPA's $PM_{2.5}$ withstood court challenges.

8.a.ii. Resolving uncertainties with data collection and creation

By 1987, the EPA had clearly recognized that a $PM_{2.5}$ standard might eventually be necessary. One notable exception to the success of the PM NAAQS review process is that the EPA did not start monitoring $PM_{2.5}$ in 1987 as a result of this suspicion. The criteria document from the 1996 PM review often bemoans the lack of monitoring data on $PM_{2.5}$. It is unclear why the EPA did not undertake to collect data on $PM_{2.5}$ starting in 1987. Perhaps it was the cost or their unwillingness to consider a $PM_{2.5}$ standard at that time for fear of industry objection invalidating their other decisions in the review. On the other hand, the PM NAAQS review process did create funding mechanisms for further PM research and the EPA accelerated their PM research starting in 1997.[49]

8.b. Influences on adaptation outcomes

Court cases and challenges to scientific justifications motivated by private interests and interest group politics played a role in determining outcomes in all four cases. For the PM

[45] Powell 1999 pg. 242 reference to endnote 5, pg. 263
[46] See Laura Johannes, "Pollution Study Sparks Debate Over Secret Data," *The Wall Street Journal*, April 7, 1997, Dockery 1993, and Pope 1995
[47] See Hillary J. Johnson, "The Next Battle Over Clean Air," *Rolling Stone,* January 2001.
[48] Pope 1995
[49] See the EPA's, "Particulate Matter Research Program," at http://www.epa.gov/pmresearch/pm_research_accomplishments/ last viewed October 2, 2006.

NAAQS, these issues only caused relatively short delays. For the SO_2 NAAQS, they were overwhelming and caused the first review to effectively last from 1979 to 2001. For the Acid Rain Program, these factors linked with a lack of salient knowledge assessments caused NAPAP to incorrectly anticipate the program's effects by missing its biggest return. Politics, although focused on the President's Clear Skies Program, which was under consideration between 2001 and 2003, also delayed the implementation of CAIR.

8.b.i. Delays in the PM NAAQS

In the first review of the PM NAAQS, two years separated CASAC's approval of the criteria document in 1982 and the EPA's initial proposal of the PM_{10} standard in 1984. As discussed in the timeline, American Steel and Iron was able to delay implementation of the PM_{10} standard by three years with a lawsuit that argued that the proposed rulemaking for PM did not "accurately reflect the latest scientific knowledge" as the CAA required.[50] Industry attempted to delay or inhibit the adoption of the $PM_{2.5}$ standard in 1996 as well. As discussed above, the PM NAAQS process also withstood this challenge. With reference to the tradeoff between the transparency of models and the challenges of consensus processes, it is interesting to note that the 1984 to 1987 delay was caused by a requirement that the EPA consider all relevant science in an open manner. In the next review, industry was able to attack the Six Cities Study because it was not transparent – the Harvard researchers did not want to share their data so that their results could be validated due to privacy concerns for the cohorts they studied.

8.b.ii. Delays in the SO_2 NAAQS

The SO_2 NAAQS suffered longer delays due to legal challenges. The legal challenges in this case again took issue with the EPA's science. But, it was the American Lung Association and the Environmental Defense Fund who brought the challenge. Industry had no incentive to spend time and money resisting the EPA's decision. First, the decision not to change the standard was in their favor because the SO_2 NAAQS was not a binding constraint on emissions due to the Acid Rain Program. Also, the public interest groups were causing delay so there was little need to worry that the standard would be reviewed again.

This long delay occurred over a relatively small issue of whether a 5-minute SO_2 standard was needed. In the mean time the SO_2 NAAQS became logically inconsistent with the other regulatory and policy programs. This delay lasted for about 22 years (from 1979 to 2001), or 27 years since a review of the SO_2 NAAQS is just now beginning in 2006. It will be interesting to see if the review that is currently starting for the SO_2 NAAQS will improve its logical consistency with the PM NAAQS, the Acid Rain Program, and CAIR.

[50] CAA §108(a)(2)

8.b.iii. The Acid Rain Program's health benefits

Although NAPAP did not explicitly recognize or estimate the potential health benefits of the Acid Rain Program before 1990, Congress was aware of the connection. For example, newspaper editorials from 1987 suggest that at least some members of the Senate believed that reducing SO_2 emissions would provide health benefits. A March 22 *Washington Post* editorial stated, "Pollution has already been cut to a point at which it no longer appears to threaten human health. Acid rain certainly kills aquatic life in lakes … but beyond that, at the levels permitted by present regulation, it's hard to show much further damage."[51] But, a senator from Maine responded to this with: "It is not true … that beyond the killing of aquatic life, 'it's hard to show much further damage.'" He then citing a hearing of the Senate environmental protection subcommittee where the American Lung Association, the American Public Health Association and the American Academy of Pediatrics "unanimously recommended that Congress take action to reduce acid rain – *based solely on health considerations.*"[52]

Enthusiastic celebration over the Acid Rain Program's success followed the initial controversy and the positive outcome gave lawmakers little incentive to update the program based on the new information about its considerable health benefits. Congress implemented the Acid Rain Program at a time when NAPAP's analyses indicated that its estimated costs overpowered its estimated benefits.[53] But, scholars and policymakers alike eventually came to see that its benefits did far outweigh its costs. [54] They also found that its novel cap-and-trade approach worked well. On the basis of benefits and costs, more stringent targets for the program were justified. The program was seen as a success and it was not until around 2003 that momentum for further reductions began to build.

8.b.iv. Clean Air Interstate Rule

The Administration first tried to update the Acid Rain Program targets based on PM health information with its Clear Skies bill, which started taking shape in 2001. Clear Skies did not pass through Congress, largely because of controversy about whether or not targets for reducing green house gasses to address climate change should be included. As discussed in the later section on the transparency of models, controversies over scientific models also added turmoil to the debate over the Administration's Clear Skies proposal.[55]

The recent implementation of the Clean Air Interstate Rule (CAIR) is a step that makes existing regulation more consistent with the science reviewed by the knowledge assessments that lead to updates the PM NAAQS. If the benefits of the Acid Rain Program

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[51] Editorial, "How Much for Acid Rain?"*The Washington Post*, 22 March 1987.
[52] George J. Mitchell, "Acid Rain: The Damage Continues," *The Washington Post*, 2 April 1987. Emphasis in original.
[53] See Portney 1990
[54] Burtraw, D., Krupnick, A., Mansur, E., Austin, D., Farrell, D., (1998) "Costs and benefits of reducing air pollutants related to acid rain," *Contemporary Economic Policy* **16**(4), 379-400 and Chestnut, L.G. and D.M. Mills, "*A fresh look at the benefits and costs of the U.S. acid rain program,*" Journal of Environmental Management **77**, 2005 pgs 252-266
[55] For a good summary of the issues see David Whitman, "Partly Sunny," *Washington Monthly*, December 2004.

are from the health benefits of reducing PM through reductions in SO_2, then CAIR is justified. However, even if one takes this as a successful adaptation of regulatory programs for SO_2 to new information, it represents a slower convergence to new science than for the PM NAAQS. NAPAP and others recognized the health information by around 1997 but the Acid Rain Program's second phase targets for 2000 were not updated accordingly. The issues surrounding the epidemiological studies that the EPA's addition of a $PM_{2.5}$ standard to the NAAQS made contentious were resolved by 2000 with the reanalysis of the Six Cities Study.

8.c. Strong incentives to surface information for PM NAAQS

Mandated reviews, interest group court cases, and successful knowledge assessment all helped push the PM NAAQS reviews forward, but after 1990 there were also strong incentives for policymakers and scientists to surface information about the health effects of $PM_{2.5}$. The Acid Rain Program created a strong incentive, external to the PM NAAQS process itself, for this information to surface.

Prior to the Acid Rain Program, there were not strong incentives for the EPA to divulge or highlight the evidence of the detrimental health effects of $PM_{2.5}$. In fact, they had reasons to avoid stressing this because of the likelihood that industry would fiercely resist a $PM_{2.5}$ standard. In their 1979 to 1987 review of the PM NAAQS, the EPA downplayed the $PM_{2.5}$ evidence by exaggerating the uncertainties around the evidence that supported choosing $PM_{2.5}$ as an indicator (see Appendix B).

After the implementation of the Acid Rain Program, the EPA, NAPAP, Congress, interest groups, and academics all had a reason to establish the health impacts of $PM_{2.5}$ and their connection to SO_2 emissions: they all wanted the Acid Rain Program to succeed. The program was Congress, the EPA, NAPAP's creation. Accolades, political capital and monetary support came with its success. The EPA and academics had supported the use of a cap-and-trade program to reduce SO_2 emissions in the Acid Rain Program because of its theoretical promise to reduce the costs of abating emissions, especially compared to traditional command and control regulations. The program's success opened doors for advances in other regulations and in academic studies. Environmental and public health groups did not initially support the cap-and-trade approach but, independently of the acid rain, they wanted $PM_{2.5}$ levels reduced.

After gaining some experience with compliance under the Acid Rain Program, industry realized that the costs of complying with a cap-and-trade program are lower than for traditional regulation, especially when initial allocations of permits were given to them freely. Then, they too had reason to desire the continued success of the Acid Rain Program so that future regulations would be modeled after it.

In 1998 Congress acted on their desire for the generation and divulgence of information about $PM_{2.5}$ by mandating and funding the EPA to accelerate research on particulate matter and the mitigation of its health effects.[56] As experience with NAPAP shows, however, large amounts of funding alone cannot guarantee that policy actions will respond to the new scientific information or that relevant work will be performed. But, the EPA now has further incentives to surface information about the relationship between SO_2 emissions and harmful PM levels because this relationship is the basis for CAIR.

8.d. Anticipation

8.d.i. Transparency of models

The U.S. cases provide some examples of when the lack of transparency in modeling or scientific processes provided ammunition for industry, or interest groups, to fight the implementation of a policy. The controversies over the Six Cities Study[57] and over CAIR (discussed below) are examples. The cases also provide one example in NAPAP where an open, consensus approach to scientific knowledge assessment led to the "watering down" of science and an ineffective knowledge assessment.[58] It is difficult to draw conclusions from these examples. Attaining the correct balance of participation and transparency on one side, and policy relevance on the other is a challenge but the examples do tend to stress the importance of transparency because whenever secrecy has been present, it has been used to cause delays.

8.d.ii. Controversy over models in Clear Skies and the Clean Air Interstate Rule

Lack of transparency in the modeling of the effectiveness of the proposed Clear Skies Program that preceded the Clean Air Interstate Rule helped public interest groups defeat Clear Skies. In an interesting twist, starting around 2002 public interest groups like the American Lung Association and Environmental Defense among others, decided not to support Clear Skies. While the root of their unhappiness with Clear Skies was the lack of targets for reductions in carbon emissions, they tried to discredit Clear Skies by claiming that the Administration was heavily influenced by industry and that Clear Skies would actually weaken the Clean Air Act targets for reducing SO_2 and NO_X.[59]

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[56] See the EPA's Particulate Matter Research webpage at http://www.epa.gov/pmresearch/ last viewed September 22, 2006.

[57] See Laura Johannes, "Pollution Study Sparks Debate Over Secret Data," *The Wall Street Journal*, April 7, 1997 and Hillary J. Johnson, "The Next Battle Over Clean Air," *Rolling Stone*, January 2001. A reanalysis of the study that confirmed its results concerns alleviated the problem, but not until 2000. See *Reanalysis of the Harvard Six Cities Study and the American Cancer Society Study of Particulate Air Pollution and Mortality*. A Special Report of the Institute's Particle Epidemiology Reanalysis Project. Final version, July 2000 at http://www.healtheffects.org/pubs-special.htm last viewed January 5, 2006.

[58] For a complete description see Herrick 2000

[59] For example, they created the website "SaveTheCleanAirAct.org" that suggested that Clear Skies would completely gut the Clean Air Act.

The ability of these groups to claim that Clear Skies would gut the Clean Air Act stemmed from a misleading EPA presentation from September 2001.[60] In a presentation to the Edison Electric Institute, the EPA tried to make the Clean Air Act look stringent so that industry would support Clear Skies. The EPA created a "straw-man" business-as-usual scenario, using modeling that was both not transparent and dishonest, which suggested that Clear Skies would be much less expensive for industry than the planned increases of stringency under the Clean Air Act. But, their characterization of the Clean Air Act was not accurate; Clear Skies would have called for emission reductions beyond those required under the CAA's Acid Rain Program. The latter called for a cap of about 9 million tons on SO_2 by 2010 and Clear Skies capped SO_2 at 4.5 million tons in 2010 and 3 million tons in 2018. Further, more credible, modeling of Clear Skies that showed that it would cause reductions was not believed after this controversy.

The flawed, nontransparent modeling in the EPA's presentation to the Edison Electric Institute provided the public interest groups with ammunition to oppose Clear Skies. This contributed to delays in adaptation of the Acid Rain Program. The EPA used the Clean Air Interstate Rule as a means to reduce the caps on SO_2 and NO_X emissions – to update the Acid Rain Program. But, this took over two years to pass after 2003 and is only germane to the eastern United States.

The models the EPA used to justify the Clean Air Interstate Rule were also controversial, but less so. The EPA states in the Preamble to the rule that models show that widespread $PM_{2.5}$ non-attainment will occur without the rule and that it will mitigate a large fraction of the non-attainment areas by 2010 and more by 2015. The EPA used air quality models to simulate base year and future concentrations of $PM_{2.5}$ with and without the regulation. The simulations include a test case that compares a back-cast for 1996 to observed data. They also include a future baseline case that estimates the extent of non-attainment if CAIR or another action were not taken. The model the EPA originally used for this process is called, "Regional Model for Simulating Aerosols and Deposition" (REMSAD). It is a photochemical grid model that uses atmospheric specie mass continuity equations.[61]

The EPA received public comments to their proposal of CAIR that REMSAD was outdated and not peer-reviewed and that 1996 was not a good base-line year because so many more monitoring data are available for later years.[62] Although the EPA states that it does not necessarily agree with these criticisms, it also used another model in response to the comments, the "Community Multiscale Air Quality Model (CMAQ)". This model is publicly available, peer-reviewed and "state-of-the-science". The EPA used 2001 data for CMAQ

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[60] This discussion is based on an extensive article on this issue: David Whitman, "Partly Sunny," *Washington Monthly*, December 2004.
[61] Details on this model are available in "Notice of Proposed Rulemaking Air Quality Modeling Technical Support Document" (NPR AQMTSD)
[62] CAIR final preamble, page 369. 40 CFR 51, 72-4,77-8, 96

tests and added seasonal calculations of air quality to its baseline analyses in response to these comments.

9. Conclusions

9.a. *A return to the questions posed in the introduction*

As a means of conclusion, return to the questions posed in the introduction that aim to understand whether certain practices at the science-policy interface made the decisions in one case more robust than those in the others.

9.a.i. Which processes have helped facilitate the assimilation and surfacing of relevant information?

In this sample of cases, the successful use of planned adaptation in the case of PM air quality helped facilitate the assimilation of new information. Comparisons of the PM case to the SO_2 air quality case and the Acid Rain Program show that many things must go right for planned adaptation to succeed. In particular, high quality knowledge assessments, incentives for information to surface, and interest group lawsuits pressuring review helped planned adaptation to succeed in the case of U.S. PM air quality standards.

A less successful knowledge assessment in the case of NAPAP resulted in severe underestimation of the benefits of the Acid Rain Program. Politically and economically motivated arguments that took the shape of disputes over science and models slowed attempts to adapt that the Acid Rain Program once the new information surfaced.

9.a.ii. Which processes reduced the nastiness of fights over the science used to justify policy options?

In all cases, there were fights over policy options that took the form of disputes over science and models. These were the most vicious in the cases of the $PM_{2.5}$ air quality standards and the Clear Skies program. The stakes in these cases were arguably higher than those in the others. The $PM_{2.5}$ standard imposed higher costs on industry than did the PM_{10} standards. Albeit retrospectively, the $PM_{2.5}$ standard has justified the Acid Rain Program and its cuts in emissions. The stakes were high for public interest groups in the case of Clear Skies; they felt that pushing the issue of carbon dioxide reductions was a large enough issue that it was worth sacrificing improvements in air quality to take a hard line on the issue of Climate Change.

Could the nastiness of fights have been reduced in these cases? In the case of $PM_{2.5}$, had the data of the Six Cities Study not been secret industry may have had less ammunition with which to question the credibility of the scientists. But, industry eventually won in the Washington D.C. District court on a legal issue of whether the EPA had overstepped its authority by creating a $PM_{2.5}$ standard. This lawsuit escalated to the Supreme Court, which upheld the EPA's $PM_{2.5}$ standard. It appears unlikely that more transparent science or models could have prevented this escalation.

Some evidence does suggest that increased transparency reduces the nastiness of fights over models and science. In the case of Clear Skies, the misleading representation of modeling on the part of the EPA contributed to fights over science. This was clearly avoidable with honest, transparent representation of knowledge. In support of the Clean Air Interstate Rule, the EPA responded early to criticism about the quality of their modeling by adopting a publicly available, peer-reviewed modeling platform. This seems to have quelled further criticisms, at least for now.

9.a.iii. Has planned adaptation enabled robust policy decisions?

There are many reasons that policies tend not to change overtime. In most cases, there are incentives for policymakers to exaggerate certainty in support of their attempts to anticipate the impacts of policies. If they then acknowledge that original policies were wrong, they will be undercutting their own credibility. Industry opposition can lead to delays in policy changes. Interest groups' desires for different or more strict standards can do the same. Political turmoil can also cause delays. Knowledge assessment processes are key in promoting change; policies can lock-in if there is no avenue through which their targets and effects can be readily and credibly reanalyzed.

The forces that typically result in the lock-in of policy decisions were present in the PM NAAQS case. For example, in 1997 EPA Administrator Browner sat before Congress and supported the 1997 $PM_{2.5}$ NAAQS by saying, "Clearly, the best available science shows that the previous standards were not adequately protecting Americans … These updated standards are based on more than 250 of the latest, best scientific studies on ozone and PM – all of them published, peer-reviewed, fully-debated and thoroughly analyzed by [CASAC]. We're talking literally peer review of peer review of peer review. It is good science. It is solid science." In addition, industry clearly did not support the adoption of a $PM_{2.5}$ standard.

The ability of the PM NAAQS program to respond to uncertainty with planned adaptation – in the face of the normal opposing forces – stems from an unusual confluence of positive influences. The NAAQS knowledge assessment process is solid. Public interest groups supported the updates to the standard and the mandated review provisions of the CAA allowed them to encourage the reviews with court cases. Additionally, there were strong external incentives for information about the health effects of $PM_{2.5}$ to surface. It was this confluence of factors that helped compel the $PM_{2.5}$ standards to change. These motivations may help explain why the U.S. $PM_{2.5}$ standards are more stringent than those in Europe.

For the other cases, the negative influences were stronger than the positive. For the SO_2 NAAQS, the Acid Rain Program and PM NAAQS diverted political and public interest.

The court case fighting the EPA's 1996 decision not to revise the SO_2 NAAQS also caused delay. Additionally, the EPA did not desire to link the SO_2 and PM standards in 1987 because that would have forced the earlier adoption of a $PM_{2.5}$ standard. For the Acid Rain Program, NAPAP's failure to address highly policy-relevant issues prior to the program's implementation was one problem. Another was that the motivation of most parties was to emphasize the program's success, not the ways that it should be altered.

Can the positive factors that promoted successful planned adaptation in the PM case be emulated in other situations? The mandating of scheduled reviews is not unique to the PM case, neither is the framework for high quality knowledge assessment. These two factors are possible to reproduce. Providing incentives for policymakers to recognize uncertainty, surface information, and change past decisions appears difficult as does ensuring that interest groups push for change rather than delay it. At the least, these cases show that these are the two big challenges to keep in mind if processes to institutionalize planned adaptation are desired.

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10. Appendix A: PM_{10} versus alternative size-specific indicators in 1987 in the US

In its 1982 Staff Paper, the EPA identified two approaches to selecting a size fraction for the PM indicator: 1) select size divisions based on how and where particles of certain sizes typically deposit in the human respiratory tract; and, 2) choose the size divisions based on typical atmospheric size distributions (1982 SP pgs. 67-8). This decision is ultimately the decision of whether to maintain the choice of a PM_{10} indicator as an option or to restrict the indicator to lie somewhere between particles with diameters less than 3 to 1 μm (PM_1 to PM_3, including $PM_{2.5}$ as an option).

They decide to focus more heavily on the first approach to determine the indicator. To justify this decision, the Staff Paper discusses the merits of each approach and the uncertainties associated with each. In doing so, the Staff Paper emphasizes the uncertainties associated with the second, rejected, approach.

The second approach of using the cut-point of the bimodal distribution of PM in the atmosphere would have entailed treating fine particles – those with diameter less than 2.5 or 3 μm – and larger modes of PM separately. This would have meant selecting a cut-point of between 1 and 3 μm. The EPA suggested that this natural dividing line was "at least partially relevant" for deposition in the human respiratory tract; studies as early as 1975 (Whitby, 1975 as cited in 1982 SP) have suggested the need to collect evidence in support of a fine particle standard (1982 SP pg. 68-9).

In support of the second approach, they noted that, although few epidemiological studies were available that studied the relationships between fine particles and mortality or morbidity, the epidemiological studies that used British Smokeshade (BS) as an indicator may be best related to the fine particles (1982 SP pg. 68-9). This conclusion draws from a study reviewed in the 1982 CD that finds that the method of measuring BS most predominantly collected particles smaller than 7-9 μm and, of those, collected particles with diameters smaller than 4-5 μm most efficiently (1982 CD pg. 103 citing McFarland *et al.* 1982). A number of epidemiological studies from the late 1960s and early 1970s found an association between PM, measured as BS, with morbidity and mortality (see 1982 CD pgs. 82-105 for a summary citing Martin and Bradley, 1960; Martin, 1964; Mazumdar *et al.*, 1981; Lawther, 1958; Lawther *et al.*, 1970).

After referencing these indications of the potential impact of fine particles on health in the 1982 SP, the EPA then lists seven uncertainties that complicate using the dividing line of PM's bimodal distribution as a cut-point for the PM indicator. Two of these uncertainties involve the fact that there is not a sharp dividing line between the fine and course PM fractions. They also note that fine and coarse particles both deposit in the tracheobronchial and alveolar regions and that size alone might not determine the health effects of the deposition of these particles; their chemical composition might be more important (1982 SP pg. 69).

The EPA emphasizes the uncertainties surrounding the option of choosing an indicator that focuses on fine PM. Several of the seven stated, and bulleted, uncertainties should logically be combined into only one uncertainty. For example, as the third uncertainty they state: "Although the two modes have differing origins and chemistries, each is chemically heterogeneous." It is striking that this point alone is not an uncertainty but a fact widely recognized at the time. The same is true for the fourth uncertainty: "The respiratory tract in effect alters the ambient distribution, with a mixture of fine and coarse particles being deposited in the tracheobronchial and alveolar regions." The fifth listed uncertainty is then: "The mixing of modes in the respiratory tract and the heterogeneity within each mode blur the distinction between the modes in terms of health effects. For example, … it is not clear that 1 µm carbon particles are more likely to result in pneumoconiosis than 4 µm silica particles" (1982 SP pg. 69). While the latter of these three 'uncertainties' is a convincing point, the three of these together logically combine to make one point, not three.

It is because of these seven stated uncertainties that the EPA chose to base its selection of an indicator primarily on where the different size fractions are deposited in the respiratory tract rather than on the bimodal distribution (1982 SP pg. 72). They were uncomfortable with the idea of using an indicator based on the size distribution, like $PM_{2.5}$ or PM_3, because it would ignore the larger particles between about 2.5 µm and 10 µm in diameter that also deposit in the respiratory tract. While the EPA highlighted the uncertainties surrounding the option of using the bimodal distribution to select a PM indicator, they did not stress any uncertainties associated with the deposition-based approach.

It may be inconsequential in a document that is a little-read as the Staff Paper, but this observation begs the question of whether the EPA stressed the uncertainty that worked toward the favored choice of the indicator for PM. A previous study of the 1979 to 1987 PM NAAQS review found through interviews that the EPA avoided a $PM_{2.5}$ standard during this review because they did not want to institute a regulatory program on SO_2 emissions through a "back-door" mechanism like the PM NAAQS (Powell 1999 pg. 242 reference to endnote 5, pg. 263). The EPA's choice of the approach to choosing the PM indicator based on where the different size fractions deposit in the respiratory tract is, however, consistent with the EPA's mandate to set health-based standards.

During the public comment period for the 1987 proposed rule to use PM_{10} as the indicator, the EPA received some comments suggesting that $PM_{2.5}$ was a more appropriate indicator because smaller particles posed greatest health concern (52 FR 24634, July 1, 1987). The EPA further explained the reasoning behind rejecting the $PM_{2.5}$ standard in its response to these comments. The logic is used is very similar to that in the SP. First, the 2.5 µm marker separates the fine and coarse fractions of PM that have different chemical and physical properties and sources. The EPA rejected the $PM_{2.5}$ indicator because of the overlap between the fine and course modes and because the respiratory tract sees the deposition of a mixture of fine and coarse mode PM in both the tracheobronchial and alveolar regions: "The mixing of these size fractions in the respiratory tract and the heterogeneity within each fraction therefore blurs the distinction between the fractions in terms of health effects" (52 FR 24634, July 1, 1987). Second, coarse dust is associated with various health problems: they believed that not all health effects come from the fine fraction alone. Also, a fine PM standard not warranted in addition to the PM_{10} standard because 1) fine PM is 40 to 70% of PM_{10} so PM_{10} standard does cover fine PM to some extent, 2) epidemiological studies do not separate the effects of fine and coarse fractions so there is not a basis for a $PM_{2.5}$ standard; and 3) it may be more "appropriate to consider the addition of chemical-specific (e.g., acid aerosols) standards rather than a fine particle standard in future primary standard revision" (52 FR 24634, July 1, 1987).

11. Appendix B: Comparison of Stringency of US and EU PM Standards

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This short piece compares the stringency of the United States air quality standards for particulate matter (PM) to the stringency of the PM standards in the European Union using data from air quality monitors in the United States and the Netherlands. According to The Pacific Research Institute (Hayward and Stowers 2004), 'it is difficult to judge whether the U.S. or the E.U. has the tougher air quality standard' since head-to-head comparisons of these norms are impossible. By using real data, a comparison *is* possible, however. In this document, the empirical data from the US is used to compare the number of monitoring stations violating annual and 24-hour standards from both sides of the ocean.[63] Data from various monitors across the United States from 2002 through 2005 were used for these calculations.[64] The empirical data from the Netherlands (period 1992 through 2004) is used to both compare the US and EU 24-hour standards to the EU annual standard and compare both unions' annual standards.[65]

Particulate matter air quality standards in the United States

There are currently three US particulate matter standards for air quality. As of September 21st, 2006 air quality in an area will meet the PM air quality standards under the following conditions:[66]

1) 24-hour PM_{10} standard: met when the 3-year average of the annual 99th percentile values for PM_{10} at each monitoring site is less than or equal to 150 $\mu g/m^3$.
2) Annual $PM_{2.5}$ standard: met when the 3-year average of the spatially averaged annual mean $PM_{2.5}$ concentrations (among designated monitors[67]) is less than or equal to 15 $\mu g/m^3$.

[63] Although the averaging times for the standards differ, we have made some pragmatic choices in order to make the comparison possible.
[64] EPA's "Air Data" at http://www.epa.gov/oar/data/ and from the EPA's "Air Quality System" at http://www.epa.gov/ttn/airs/airsaqs/detaildata/downloadaqsdata.htm last viewed October 1, 2006.
[65] The Dutch data derive from the national air quality monitoring network (Landelijk Meetnet Luchtkwaliteit, LML) operated by the Netherlands National Institute for Public Health and the Environment (RIVM). Website (with downloadable data): http://www.rivm.nl/milieukwaliteit/lucht (in Dutch).
[66] 40 CFR Part 50

3) 24-hour $PM_{2.5}$ standard: met when the 3-year average of the annual 98th percentile values for $PM_{2.5}$ at each monitoring site is less than or equal to 35 $\mu g/m^3$.

Prior to this recent decision, the PM air quality standards in the US also included an annual PM_{10} standard that was met when the 3-year average of the annual mean PM_{10} concentrations at each monitoring site were less than or equal to 50 $\mu g/m^3$. The EPA's 2006 decision also lowered the 24-hour $PM_{2.5}$ standard from 65 $\mu g/m^3$.

Particulate matter air quality standards in the European Union

The air quality standards in the EU include

1) Annual PM_{10} standard: met when the annual average PM_{10} value is less than 40 $\mu g/m^3$.
2) 24-hour PM_{10} standard: met when the 24-hour average values of PM_{10} exceed 50 $\mu g/m^3$ fewer than 36 times.
3) Proposed $PM_{2.5}$ standard: met when the annual average $PM_{2.5}$ value is less than 25 $\mu g/m^3$.

The US annual $PM_{2.5}$ standard of 15 $\mu g/m^3$ is clearly more stringent than the proposed EU standard of 25 $\mu g/m^3$ even with three-year and spatial averaging. The PM_{10} standards are more difficult to compare without data. Below we present two original comparison exercises based on US and Dutch data.

Comparison of PM_{10} standards, part I: Exceedences at the monitoring site level in the United States in 2005

We here first present the results of the comparison exercise based on the US data. Calculation of compliance with the US standards requires three years of data for all standards (to calculate compliance in 2005, data from 2002 through 2005 are needed). The similar calculations for the EU standards only require data from the year in question. These calculations are all for compliance in 2005. For a set of 989 monitoring sites across the US, table 1 and figures 1 and 2 below summarize the comparison results.

Number of exceedences for 2005			
U.S. 24-hour	EU 24-hour	U.S. Annual	EU Annual
49	47	18	54

Table 1

Number of US monitoring sites that did not meet the standards in 2005.

[67] Designated monitors are specifically designated for spatial averaging as allowed under 40 CFR Part 58.

From table 1 we can conclude that the EU and US 24-hour PM_{10} standards are of similar stringency and that the EU annual standard is definitely more stringent than the – now defunct – annual standard for PM_{10} in the United States.

Figure 1 shows us that the exceedences of the two 24-hour standards are not well correlated. Of the 49 sites that did not meet the US 24-hour PM_{10} standard and the 47 that did not meet the EU standard, only 20 did not meet both.

Figure 2 gives an indication of the significant year-to-year variability in the annual mean PM_{10} concentration for each monitoring station. For instance, several monitors that were in compliance with EU norms in 2005, were not in compliance with US norms over the 2002–2005 averaged period.

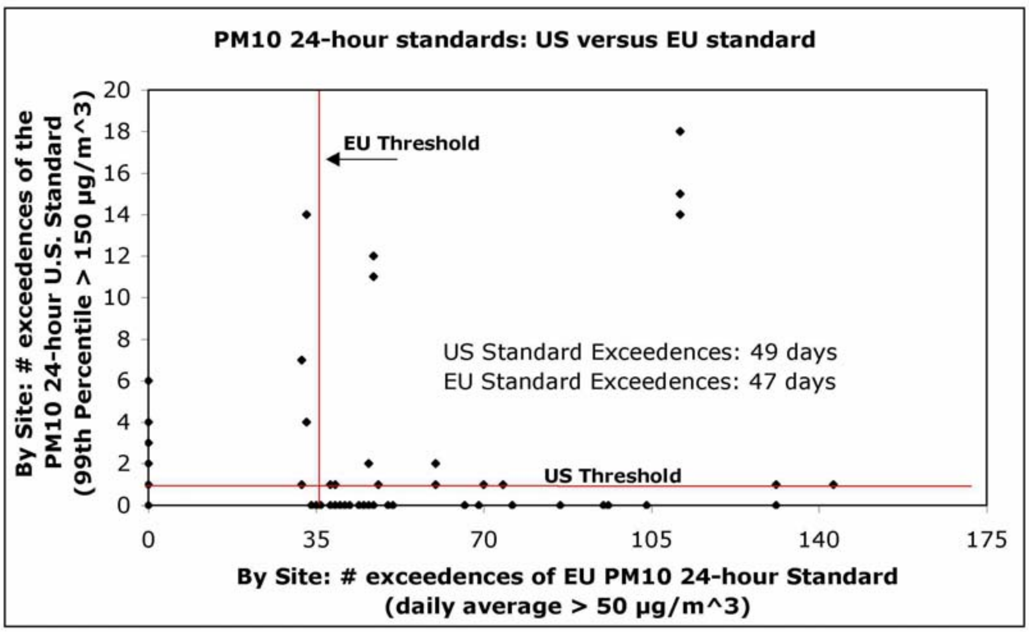

Figure 1

Exceedence of US and EU 24-hour standards for each US site in 2005. For each site, the number of days the US standard of 150 μg/m³ is exceeded (vertical axis; calculation based on a running average) and the number of days the EU standard of 50 μg/m³ is exceeded (horizontal axis) are shown. Note that 913 of the US sites do not show an exceedance of any of both 24-hour standards. All those sites are represented by the single dot in the origin. [read “sites” for “days” in the figure]

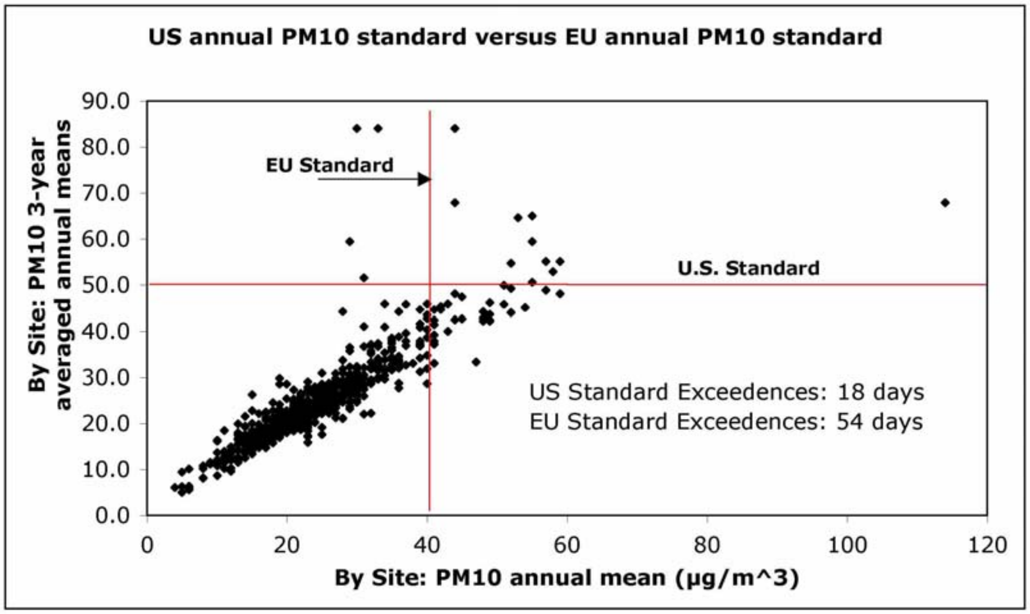

Figure 2

Exceedence of US and EU annual standards for each US site in 2005. For each site, the 3-year averaged annual mean (vertical axis) and the 2005 annual mean (horizontal axis) are shown. [read “sites” for “days” in the figure]

Comparison of PM_{10} standards, part II: Exceedences based on spatially averaged data from the Netherlands 1992–2004

In the second comparison exercise, daily values of PM_{10} over the period 1992 through 2004 were used from 18 stations distributed over the Netherlands. All data for January 1st were removed because of the high concentrations due to fireworks. This effect is not what we are specifically interested in. Of the 18 stations, nine are rural background, four are urban background and five are urban street stations. The results are shown in figures 3, 4 and 5.

From figure 3 we can conclude that more exceedences occurred of the 24-hour EU standard than of the annual EU standard. The assumption behind the 1999 Daughter Directive that introduced both standards was that the two limit values for PM_{10} were equivalent; based on the knowledge at that time they were they thought to be equally ‘stringent’. In practice, this has turned out not to be the case. The limit value for the 24-hour average is more ‘stringent’ than that for the annual average concentration (limit of 40 $\mu g/m^3$). The EU daily norm is equivalent to a yearly average limit of about 31 $\mu g/m^3$. What is remarkable in this figure is that the spread around the green trend line is small. This means that in Netherlands one norm would suffice instead of two. This can either be a daily or an annual average norm. A choice could be: an annual average PM_{10} norm of 31 $\mu g/m^3$, which – coincidentally – is near the norm of 33 $\mu g/m^3$ proposed by the European Parliament on September 26, 2006.

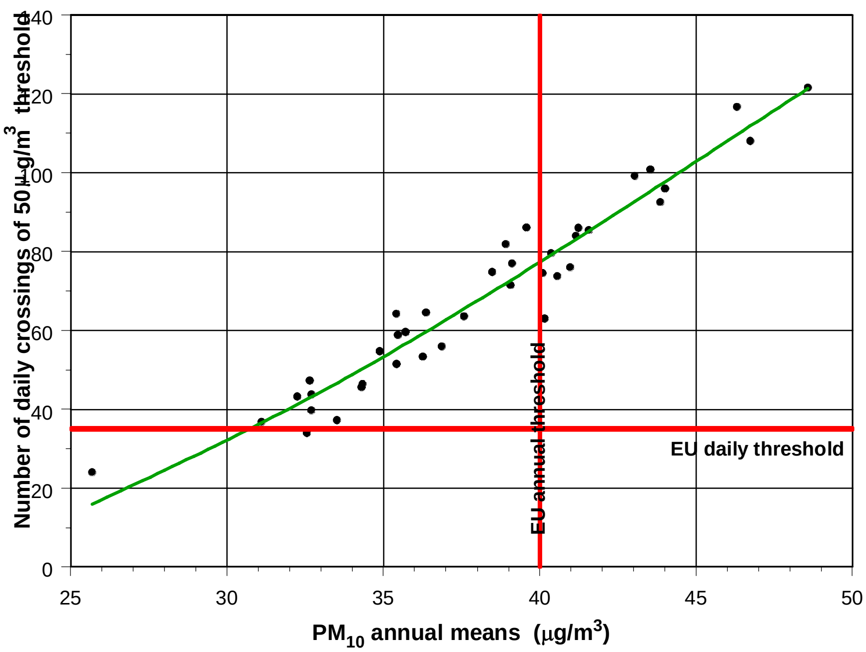

Figure 3

PM_{10} annual means (horizontal axis) and number of daily crossing of EU norm of 50 µg/m³ (vertical axis) for 18 stations distributed over the Netherlands over the period 1992–2004. The EU daily and annual thresholds are indicated by red lines. Each dot shows an annual mean for a particular year averaged of all stations of the same type (rural background: nine stations; urban background: four stations; and urban streets: five stations).

Folkert et al. (2005, 34) mention that the proposed concentration cap of 25 µg/m³ for $PM_{2.5}$ is derived from the annual averaged limit value for PM_{10} of 40 µg/m³ using a factor 0.6. Since the ratio of $PM_{2.5}/PM_{10}$ is estimated to be about 0.75 (0.6–0.85) in the Netherlands, the $PM_{2.5}$ concentration cap is stricter than the PM_{10} annual limit value and matches a yearly averaged PM_{10} concentration of about 33 µg/m³. Thus the daily limit value for PM_{10} of about 30 µg/m³ will remain the strictest limit. This may become different if the $PM_{2.5}$ standards of 20 µg/m³ proposed by the European Parliament on September 26, 2006 is endorsed.

In figure 4, the US daily norm is compared to the EU annual norm. It appears that the US daily norm is approximately equivalent to an annual average norm of 42 µg/m³. The US daily norm is thus somewhat less stringent than the EU annual average norm. Still, the scatter around the green line is large. Thus setting a daily norm is really different from setting an annual average norm.

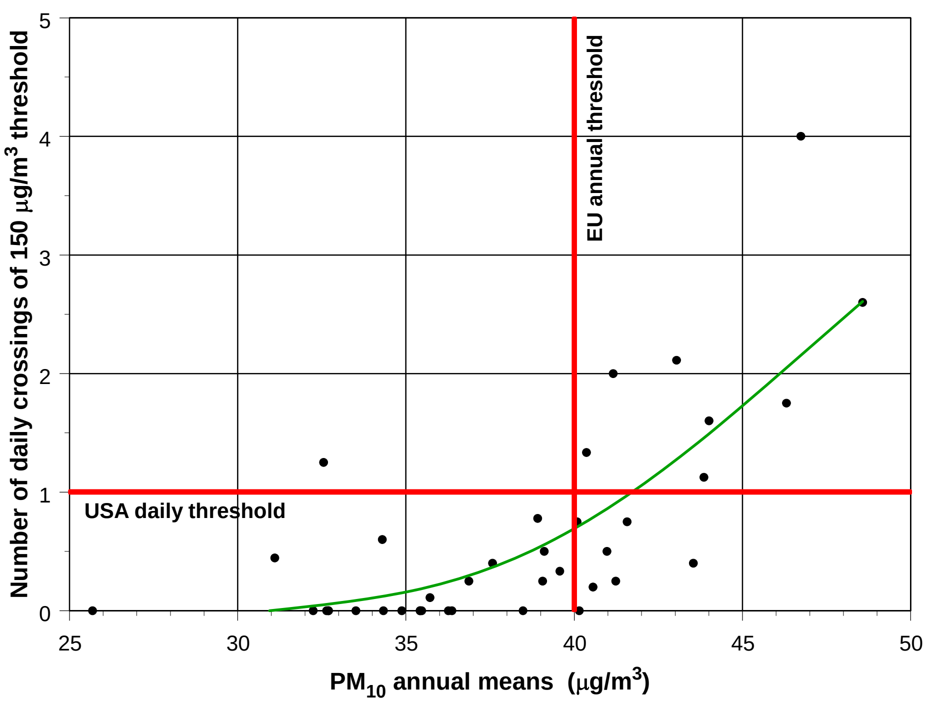

Figure 4

PM_{10} annual means (horizontal axis) and number of daily crossing of US norm of 150 μg/m³ for 18 stations distributed over the Netherlands over the period 1992-2004. The EU annual and the US daily threshold are indicated by red lines. For the definition of the dots, see figure 3.

Finally, in figure 5, as in figure 2, the US and EU annual limits are compared. This figure shows that 50 microgram smoothed over 3 years does never occur in the Netherlands. Conversion of the US norm into a non-smoothed threshold (commensurable to the EU annual average measure) is thus unreliable (it is risky to extrapolate the EU annual average measure) is thus unreliable (it is risky to extrapolate the green line, a flexible trend estimate). But at least it may be concluded from figure 5 that the EU norm is stricter. This corroborates the results for the US data.

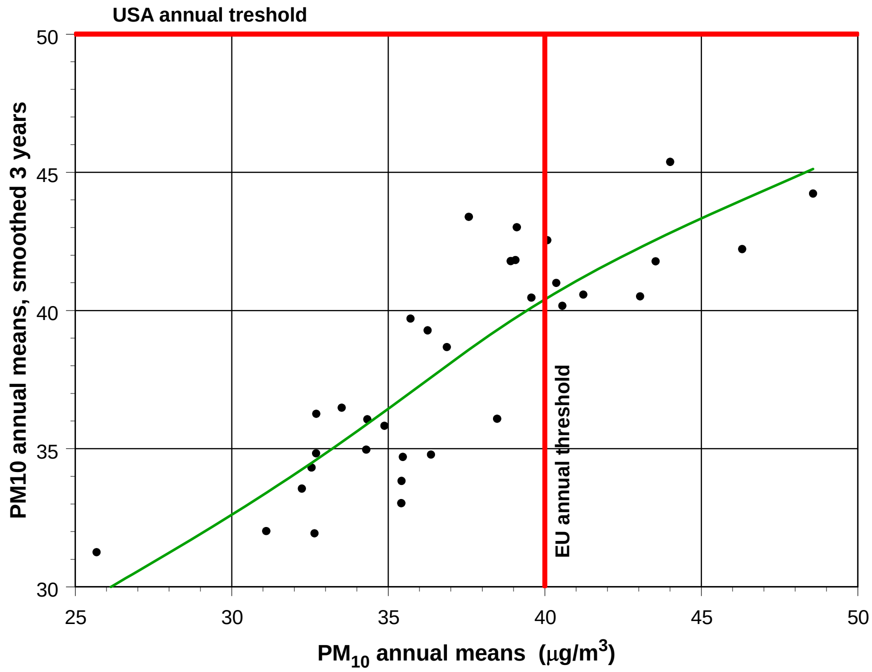

Figure 5

PM_{10} annual means according to the EU definition (horizontal axis) and the US definition (vertical axis) for 18 stations distributed over the Netherlands over the period 1992-2004. The EU and US annual thresholds are indicated by red lines. For the definition of the dots see figure 3.

References

Folkert et al., R.J.M. et al. (2005), *Consequences for the Netherlands of the EU Thematic Strategy on Air Pollution*, MNP Report 500034002, Bilthoven: Netherlands Environmental Assessment Agency (MNP).

Hayward, S.F and Stowers, R. (2004), 'Air quality: The U.S. and Europe compared', in *2004 Index of Environmental Indicators*, San Francisco: Pacific Research Institute. Available at: http://www.pacificresearch.org/pub/sab/enviro/04_enviroindex/09_air.html.